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Cover: Tail of the eastern Northern Pacific Gray Whale. Photo courtesy of Jim Darling.

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Monitoring the Endangered Species Act: Revisiting the Eastern North Pacific Gray Whale

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Abstract

The U.S. Endangered Species Act provides powerful legislation to conserve imperiled populations but provides little consideration for the long-term viability of species that are deemed "recovered" and subsequently removed from the ESA List. Since it's inception in 1973, a mere 15 species have been delisted from the ESA (Noecker 1998). The eastern north Pacific gray whale was the first marine mammal to be removed from the ESA and has been applauded as an endangered species "success" story (Gerber et al. 1999). In recent years, gray whales have experienced a steady population decline – in fact, the most recent abundance estimate suggests that the population has declined by 33% since delisting (Rugh 2003). In light of this population decline, in this paper we re-examine the risk of extinction and ESA status of eastern north Pacific gray whales. We determined that the current population decline does not warrant reclassification as threatened or endangered, but that longer timeseries are needed to obtain a realistic picture of population dynamics. Given the uncertain trajectory of delisted species, monitoring beyond the 5 years required by the ESA is needed to ensure long-term viability of species removed from the list.

Resumen

La ley de especies en peligro de extinción (ESA en Ingles) contiene legislación que facilita la conservación de las especies en peligro, pero presta poca atención a la viabilidad a largo plazo de las especies consideradas como "recuperadas" y removidas de la lista de especies en peligro de extinción (ESA). Desde sus inicios en 1973, tan solo 15 especies han sido removidas de la lista (Noecker 1989). La ballena gris del noreste en el océano Pacifico fue el primer mamífero removido de la lista ESA y considerada como éxito en la recuperación de especies (Gerber et al. 1999). Sin embargo, en años recientes las poblaciones de ballenas grises se han reducido constantemente; de hecho, las últimas estimaciones de la abundancia de sus poblaciones han declinado en un 33% desde que fueron removidas de la lista (Rugh 2003). En vista de la declinación de la población, en este articulo re-examinamos el riesgo de extinción y el status de la ballena gris del Pacifico en la lista ESA. Determinamos que la reciente declinación de la población no justifica su reclasificación como especie amenazada o en peligro de extinción, pero que se necesitan estudios de más larga duración para obtener una mejor imagen de su dinámica poblacional. Dada la incierta trayectoria de especies removidas de la lista, se hace necesario el monitoreo por mas de los cinco años requeridos por ESA para asegurar la viabilidad a largo plazo e especies removidas de la lista.

Introduction

Does delisting species under the Endangered Species Act (ESA) ensure that a species will remain viable in the foreseeable future? Because few species have been removed from the ESA, we have little empirical evidence from which to gauge our success in the conservation of endangered or threatened populations. Currently, about 1,800 species are protected under ESA regulations, and a mere 39 species have been removed from the list to date. Furthermore, of the 39 delisted species, only 15 species are considered recovered (Table 1). Current ESA provisions specify a 5-year monitoring period following delisting (U.S. Endangered Species Act, 1973) but consideration for long-term viability is lacking. The eastern North Pacific gray whale (*Eschrichtius robustus*) provides us with an interesting case study since it was the first marine mammal to be removed from the ESA and continuous monitoring has been conducted since its delisting in 1994 (Gerber et al. 1999). The population exhibited an increase in abundance during its protection under the ESA and during the 5-year monitoring period following its delisting (Figure 1). Ironically, abundance estimates following this 5-year period indicate a consistent decline in abundance - by 2002 the population had declined to 17,500 from 26,635 estimated in 1998 (Rugh 2003). Cumulatively, this decline represents a loss of 10,000 animals, constituting 1/3 of the total population estimated in 1998 (Rugh 2003). However, more recent work brings into question the significance of this decline and indicates other possible explanations for the low abundance estimate in 2002 (Anonymous 2004a, 2004b). The specific cause of this decline has not been determined but may be related to a change in carrying capacity (e.g., a decrease in prey species in Alaskan feeding grounds, Rugh 2003).

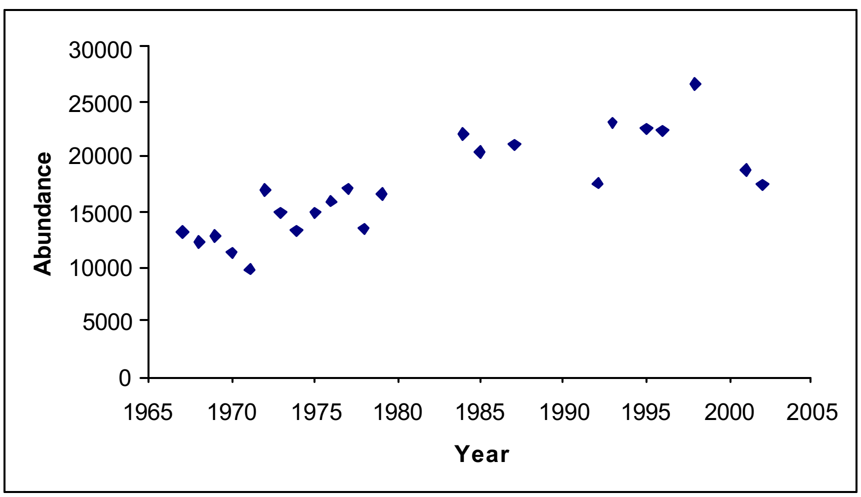

Figure 1. Abundance data for the eastern North Pacific Gray Whale. Census data during the period in which the population was listed under the ESA (1970-1994) show an increase in abundance. Post delisting abundance estimates indicate a decline following the five-year monitoring period (1994-1999).

Changes proposed to ESA policy in 2003 address the absence of long-term monitoring programs for delisted species but indicate that the current focus is on recovery programs for listed species (Federal Register 2003). These regulations do however indicate that future threats to declining populations must be taken into account before the threat is actually impacting the target species. Threats to delisted species are not specifically addressed in regulation and since few of the species removed from the ESA actually occur in areas within U.S. jurisdiction it is often difficult to predict future threats since they are not in our "backyard." Fortunately, many of our delisted species are protected by other pieces of legislation. For example, gray whales fall under the auspices of the International Convention for the Regulation of Whaling and the U.S. Marine Mammal Protection Act of 1972. Other legislation such as the U.S. Migratory Bird Treaty Act of 1918 and the Clean Water Act also provide limited protection for species that are removed from the ESA. These background laws do provide some measure of protection but some argue that until ample post delisting protection is available, maintaining species on the list of threatened and endangered wildlife may be the best approach to ensure the viability of an imperiled species until further legislation is enacted (Doremus 2001).

Conservation biologists are often faced with limited data concerning the fate of declining species, but must make policy decisions in the face of limited information (Doak and Mills 1994). Having time-series data of abundance estimates over 15 years is rare for long-lived vertebrates and even more rare for endangered species (Gerber et al. 1999). The value of having a long time-series of monitoring data for threatened or endangered populations has been demonstrated in other studies, which indicate that longer lengths of census data provide less uncertainty in listing recom-

Date Species First Listed	Date Delisted	Species Name	Reason Delisted
03/11/1967	06/04/1987	Alligator, American (*Alligator mississippiensis*)	Recovered
11/06/1979	10/01/2003	Barberry, Truckee (*Berberis (=Mahonia) sonnei*)	Taxonomic revision
02/17/1984	02/06/1996	Bidens, cuneate (*Bidens cuneata*)	Taxonomic revision
08/27/1984	02/23/2004	Broadbill, Guam (*Myiagra freycineti*)	Believed extinct
04/28/1976	08/31/1984	Butterfly, Bahama swallowtail (*Heraclides andraemon bonhotei*)	Act amendment
10/26/1979	06/24/1999	Cactus, Lloyd's hedgehog (*Echinocereus lloydii*)	Taxonomic revision
11/07/1979	09/22/1993	Cactus, spineless hedgehog (*Echinocereus triglochidiatus* var. *inermis*)	Not a listable entity
09/17/1980	08/27/2002	Cinquefoil, Robbins' (*Potentilla robbinsiana*)	Recovered
03/11/1967	09/02/1983	Cisco, longjaw (*Coregonus alpenae*)	Extinct
03/11/1967	07/24/2003	Deer, Columbian white-tailed Douglas County DPS (*Odocoileus virginianus leucurus*)	Recovered, threats removed
06/02/1970	09/12/1985	Dove, Palau ground (*Gallicolumba canifrons*)	Recovered
03/11/1967	07/25/1978	Duck, Mexican (U.S.A. only) (*Anas "diazi"*)	Taxonomic revision
06/02/1970	08/25/1999	Falcon, American peregrine (*Falco peregrinus anatum*)	Recovered
06/02/1970	10/05/1994	Falcon, Arctic peregrine (*Falco peregrinus tundrius*)	Recovered
06/02/1970	09/12/1985	Flycatcher, Palau fantail (*Rhipidura lepida*)	Recovered
04/30/1980	12/04/1987	Gambusia, Amistad (*Gambusia amistadensis*)	Extinct
04/29/1986	06/18/1993	Globeberry, Tumamoc (*Tumamoca macdougalii*)	New information discovered
03/11/1967	03/20/2001	Goose, Aleutian Canada (*Branta canadensis leucopareia*)	Recovered
10/11/1979	11/27/1989	Hedgehog cactus, purple-spined (*Echinocereus engelmannii* var. *purpureus*)	Taxonomic revision
12/30/1974	03/09/1995	Kangaroo, eastern gray (*Macropus giganteus*)	Recovered
12/30/1974	03/09/1995	Kangaroo, red (*Macropus rufus*)	Recovered
12/30/1974	03/09/1995	Kangaroo, western gray (*Macropus fuliginosus*)	Recovered
06/02/1977	02/23/2004	Mallard, Mariana (*Anas oustaleti*)	Believed extinct
04/26/1978	09/14/1989	Milk-vetch, Rydberg (*Astragalus perianus*)	Recovered
06/02/1970	09/12/1985	Owl, Palau (*Pyroglaux podargina*)	Recovered
06/14/1976	01/09/1984	Pearlymussel, Sampson's (*Epioblasma sampsoni*)	Extinct
06/02/1970	02/04/1985	Pelican, brown (U.S. Atlantic coast, FL, AL) (*Pelecanus occidentalis*)	Recovered
07/13/1982	09/22/1993	Pennyroyal, Mckittrick (*Hedeoma apiculatum*)	discovered
03/11/1967	09/02/1983	Pike, blue (*Stizostedion vitreu m glaucum*)	Extinct
10/13/1970	01/15/1982	Pupfish, Tecopa (*Cyprinodon nevadensis calidae*)	Extinct
09/26/1986	02/28/2000	Shrew, Dismal Swamp southeastern (*Sorex longirostris fisheri*)	New information discovered
03/11/1967	12/12/1990	Sparrow, dusky seaside (*Ammodramus maritimus nigrescens*)	Extinct
06/04/1973	10/12/1983	Sparrow, Santa Barbara song (*Melospiza melodia graminea*)	Extinct
11/11/1977	11/22/1983	Treefrog, pine barrens (FL pop.) (*Hyla andersonii*)	New information discovered
09/13/1996	04/26/2000	Trout, coastal cutthroat (Umpqua R.) (*Oncorhynchus clarki clarki*)	Taxonomic revision
06/14/1976	02/29/ 1984	Turtle, Indian flap-shelled (*Lissemys punctata punctata*)	Erroneous data
06/02/1970	06/16/1994	Whale, gray (except where listed) (*Eschrichtius robustus*)	Recovered
03/11/1967	04/01/2003	Wolf, gray U.S.A. (*Canis lupus*)	Taxonomic revision
07/19/1990	10/07/2003	Woolly-star, Hoover's (*Eriastrum hooveri*)	New information discovered

Table1. Species that have been removed for the list of Threatened and Endangered Wildlife, indicating date listed, date delisted and reason for delisting. A total of 39 species has been removed from the ESA but only 15 are considered recovered.

mendations (Gerber et. al 1999). While listing criteria developed for long-lived vertebrates use population viability analysis (PVA) models to determine listing recommendations, it is difficult to identify how much data is appropriate for PVA analysis and in many cases data may not be adequate to obtain reliable listing recommendations (Brook and Kikkawa 1998). Generally, data are unreliable or many times simply unavailable (Caughley and Gunn 1996). The extensive data available for the eastern Northern Pacific (ENP) gray whale provides us with a pertinent case study for identifying the role of post-delisting monitoring data in endangered species recovery. In this study we apply recently

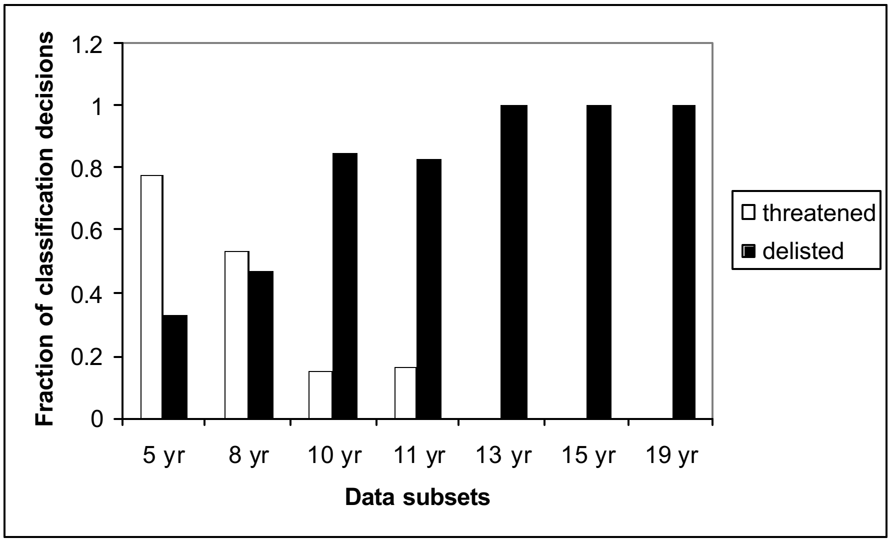

Figure 2. Classification decisions for data subsets. Data subsets containing thirteen years or more of census data unequivocally support a recommendation to keep the gray whale delisted from the ESA while smaller subsets yielded ambiguous listing recommendations.

developed quantitative criteria for deciding how to classify species (i.e., as endangered, threatened, or delisted, Gerber and DeMaster 1999) to the recently delisted gray whale.

Methods

I. Estimating Growth Rate Values

Using census data collected annually by the National Marine Fisheries Service (NMFS), we used a simple diffusion-approximation model (Dennis et al. 1991) to estimate the growth rate of the eastern North Pacific gray whale population. Since previous assessments failed to identify density dependence in gray whales (Gerber et al. 1999), we use an exponential model:

$$N_{t+1}=\lambda_t N_t$$

where N is the population and λ is the growth rate for year t. The model assumes that population variability is caused by environmental stochasticity and density-dependent variables that may influence population growth are not a factor. In light of uncertainty associated with density dependence in gray whales, this model is robust to violations of the density dependent assumption (Sabo et al. 2004).

II. Listing Criteria

The approach for species classification developed by Gerber and DeMaster (1999) for long-lived vertebrates focuses on three aspects of a population; population abundance and the tendency for this population to increase and decrease over time (μ) as well as variablility of growth rates (σ^2). The method developed by Gerber and DeMaster (1999) allows us to use the growth rate estimates from our diffusion approximation model and incorporate sampling error inherent in the fluctuating population. This approach is based on a probability-driven model of population demographics, which establishes threshold levels for threatened and endangered status by projecting a growth rate back from a specified quasi-extinction level (N_q). To determine endangered status, the population would have to have a >5% chance of falling below N_q (500 individuals; Best 1993) during the next 10 years. For the status of threatened, the population would have to have greater than a 5% chance of falling below the threshold level in the next 35 years. If the population remains above the threshold in both cases, the species should be considered for delisting from the ESA.

To examine the importance of post-delisting monitoring data on ESA listing decisions, we examined a variety of data subsets using abundance information on the eastern North Pacific gray whale. Following Gerber et al. (1999), data subsets ranged from 5, 8, 10, 11, 13, 15 and 21-year samples. For example, for 5-year intervals there were 18 possible combinations of data, and for 8-year intervals there were 15 combinations of data. Five-year samples were particularly relevant to illustrating the plausible consequences of the 5 years of monitoring mandated by the ESA on classification decisions.

Results and Discussion

Gerber et al. (1999) found that a quantitative decision to delist was unambiguously supported by eleven years of data, but precariously uncertain with fewer than ten years of data. Interestingly, the decision to delist is robust to the recent decline of 34% between 1997 and 2002. In particular, the application of the Gerber and DeMaster (1999) approach to current abundance data for the eastern North Pacific gray whale did not support a decision to reclassify this species as endangered or threatened. However, in light of the recent population decline, thirteen (vs. eleven) years of data are needed to unequivocally support a decision to delist (Figure 2). Furthermore, although the population has been declining at a rate of 34% for the last 4 years, the population growth rate needed to warrant reclassification as endangered is 0.881 and 0.905 for reclassification as threatened.

Our results convey the importance of analysis of post delisting monitoring data. If abundance estimates were not being conducted by the NMFS, the current decline may have gone unnoticed. There is a great deal of uncertainty about whether the recent decline is an acute event or an ongoing situation (Rugh 2003). Other large whale stocks may have recovered to the point where they may become delisted in the near future but without long-term monitoring programs, the fate of these species is uncertain. The development of recovery programs for listed species has become the main focus of the ESA. Our results emphasize the importance of including a long-term monitoring plan in the recovery program to ensure continued viability of listed species.

The effectiveness of the ESA has been evaluated by Doremus and Pagel (2001) who view the limited number of delisted species as a strength of the Act. While it may be beneficial for many imperiled populations to remain listed and benefit from the ESA's legal protection, this raises challenges in recovery programs for an increasing list of threatened and endangered species. Gerber (2003) points to the fact that without successful recovery stories there may be little political support for the ESA given its current track record. Furthermore, while the ESA may have been successful at preventing extinction for numerous species, promoting recovery (vs. preventing extinction) may be a more appropriate conservation goal for listed species. As the ESA develops in the future these and other issues need to be addressed with provisions for long-term monitoring being of high priority.

Monitoring of delisted species is only part of the formula we need to ensure the fate of these populations. The removal of current or potential threats is also an integral component. Of the recovered species taken off the ESA, overharvesting was the greatest threat to five of these recovered species that includes the ENP gray whale (Noecker 1998). It may be easy to enforce "no-take" legislation, but when a species is affected by factors such as environmental stochasticity, it is difficult or impossible to remedy these situations. One strength of collecting population data on a species is the ability to observe changes in a population before a crisis situation is obvious. This is demonstrated with the ENP gray whale example beacuse it was monitored biannually by the National Marine Fisheries Service for 6 years following delisting. However, given the difficulty in detecting declines on the order of 1-5% per year for populations with significant uncertainty in abundance estimates (Gerrodette 1987), we recommend consideration of longer time periods for post-delisting monitoring. Unfortuantely, current funding levels and associated proirities from Congress are such that future monitoring of the population is in question. It appears that Congress will have to provide specific advice and funding to the agencies responsible for implementing the ESA; otherwise the potential for suitably protecting apparently recovered populations will be lost.

The ESA has the ability to protect species ranging from small subalpine plants to large megafauna around the globe but each individual species listed under the ESA may require specialized monitoring programs. It will be difficult to porvide suitable rationale for a fixed period of post-delisting monitoring that works for all species, but efforts should be made to evaluate the merits of monitoring requirement on the order of 5 to 10 years. A lack of data may lead to inappropriate decisions and inconsistent management of imperiled species (Tear et al. 1993). Long term monitoring of delisted species can provide us with the information needed to determine if a population is "recovered" or if anthropogenic or environmental factors continue to affect long-term viability. The ability of the ESA to protect a target species has not been clearly demonstrated and monitoring provides us with a gauge for our successes and failures in the management of endangered and threatened populations.

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Population Viability Analysis: Theoretical Advances and Research Needs

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Abstract

Population viability analysis (PVA) is a set of tools for forecasting population growth and estimating extinction risk. Recent methdological advances include assessing model reliability by estimating model parameters from time series with observation errors, synthesis of PVA with decision theory, extension of models to sex-structured populations and to density dependence in age-structured populations, and experimental validation of qualitative model predictions. Future research should focus on developing biologically based models of population growth, model selection, and model averaging; extension of models to species with complicated life histories including quiescent stages and seasonal life histories; and developing tools for validly incorporating additional information about species' demography from knowledge of their ecologies and natural histories. PVA is a rapidly developing methodology and users should recognize that a large variety of models and techniques are available for population forecasting.

Resumen

El análisis de la viabilidad de poblaciones (PVA-en ingles) es un juego de herramientas para pronosticar el crecimiento de las poblaciones y para estimar su riesgo de extinción. Avances metodológicos recientes incluyen la evaluación de la validez de los paramentos de modelos de « time series » con errores de observación, síntesis del análisis de viabilidad poblacional con la teoría de decisiones, la extensión de modelos a poblaciones estructuradas por sexo con poblaciones estructuradas por edad, y la validación experimental de modelos cualitativos de predicción. Modelos futuros de investigación deberían enfocarse en el desarrollo de modelos biológicos de crecimiento poblacional, la selección de modelos, el promedio de modelos, la extensión de modelos a especies con historias de vida complejas incluyendo historias de vida temporales y etapas de quiescente, y el desarrollo de herramientas de validación incorporando información adicional sobre la demografía de especies usando conocimiento de su ecología y su historia natural. El PVA es un método en rápido desarrollo y los usuarios deben darse cuenta que disponen de una larga lista de modelos y tecnologías para la predicción de poblaciones.

Population Viability Analysis (PVA) arose in the 1980's as a set of tools for conservation biologists to determine the risks faced by threatened and endangered species (Soulé 1987, Shaffer 1981). Development of PVA was prompted by the emerging extinction crisis and especially by concern for three endangered North American species—Grizzly bear (*Ursus Arctos*), Northern spotted owl (*Strix occidentalis*), and Loggerhead sea turtle (*Caretta caretta*), all of which were featured in early applications of PVA. During this period, important theoretical developments from the 1970's and early 1980's were widely publicized and the first case studies were conducted.

In the early 1990's two important developments occurred. First, Dennis *et al*. (1991) demonstrated how the parameters of the stochastic exponential population growth process could be estimated with regression analysis and how error in these estimates could be propagated through model equations to estimate uncertainty in the chance of extinction. The results from propagating sampling errors are now sometimes called population projection intervals (Lande *et al.* 2003) and are crucial if management actions are to be adequately informed. Second, software for simulating population trajectories on desktop computers became widely available (Lacy 1993). Notwithstanding more recent developments, especially for the analysis of metapopulations and for nonlinear models of population growth, the combination of the stochastic exponential population growth models and publicly available simulation software probably still represent the majority of PVA's being conducted now, a decade later.

The apparent simplicity of PVA, especially when it relies heavily on these two techniques, has repeatedly raised concern that it will be used incorrectly when deciding conservation actions. Recently, Reed *et al.* (2002) reviewed four well-known and common shortcomings of PVA models: (1) Models generally do not consider the spatial distribution of individuals in the population. (2) The practical consequences of uncertain parameter estimates are generally not sufficiently explored through sensitivity analysis. (3) Genetics is rarely considered, even though the deleterious effect of inbreeding on individual fitness is well established. And, (4) the appropriate models for plants are poorly understood compared to models for vertebrates. Importantly, methods to accommodate each of these obstacles exist and are outlined and advocated by Reed *et al.* (2002) as well as in other introductions to PVA (*e.g.* Morris and Doak 2002). There is no reason therefore for future PVAs to overlook these issues, and one hopes that our confidence in model predictions will increase commensurately.

Additional areas of active research are the reliability of model forecasts (Ludwig 1999, Fieberg and Ellner 2000), the synthesis of PVA and tools for structured decision analysis (Possingham *et al.* 2002, Ludwig and Walters 2002), density-dependence in age-structured stochastic populations (Lande *et al.* 2002), experimental validation of model predictions (Drake and Lodge 2004, Belovsky *et al.* 1999), and sex-structured models (Lande *et al.* 2003). No doubt, I have overlooked some topics and still others will be forthcoming. In this active stage of research therefore it is imperative that the practice and theory of PVA develop simultaneously. So, in conclusion, I identify three areas in which research is most desperately needed.

First, PVA's would be greatly improved by the development of more realistic, nonlinear models of population growth, where model forms are derived from the underlying biology of the population growth process, rather than because of phenomenological fit. It is well known that nonlinearity can have important consequences for the predictability of population dynamics, even over very short periods of time. The assumption that population growth is exponential at small sizes is therefore unwarranted, especially among sexually reproducing species or species whose survival depends on cooperation among individuals (Lande *et al.* 2003). Of course, once a set of models has been developed and is reasonably well understood, it still remains to decide which models to use for forecasting (Burnham and Anderson 1998). Finally, it will often be the case that a single model is not obviously best when compared with observed data. Under such considerations, methods for averaging model forecasts will be required if predictions are to be reliable (Burnham and Anderson 1998). This is presently an active area of research in statistics (*e.g.*Hoeting *et al.* 1999) though these techniques have rarely been brought over to conservation biology.

Second, PVA methods should be extended for species with complicated life histories. The standard models allow for age-structure, which is important, but generally assume that population growth is not density-dependent and therefore asymptotically exponential. This is the basis of the Leslie/Lefkovitch matrix formulation of structured population growth. See Caswell (2001), chapters fourteen and fifteen, for an introduction to this topic. Lande *et al.* (2002) have extended the concept of density-dependence to age-structured populations, however other issues remain. Many species, especially non-vertebrates, exhibit quiescent life stages of indeterminate duration. Other species exhibit periodical life histories, while still others exhibit life histories that are determined by environmental conditions. A particularly challenging problem for these species is how to obtain precise and unbiased estimates of parameter values for stages that are not easily observed. This problem is exacerbated when population dynamics are confounded by large sampling error, sometimes the result of extremely large population sizes, *e.g.* seed production wherein every seed constitutes an individual organism, or heterogeneous environments.

Finally, we need better ways of incorporating natural history data when estimating model parameters. Sometimes, the biology of species for which PVA are conducted is very well understood because the species is of great conservation interest. But, because the more familiar techniques for estimating growth rates and variances depend on time series, parameter estimates are wildly uncertain because time series are short. Unfortunately, the timeframe for accurate forecasts attenuates rapidly with parameter uncertainty (Ludwig 1999, Fieberg and Ellner 2000). Therefore, especially when background information is available pertaining to such important factors as the frequency of disturbance events, rates of development, inter-specific interactions, mating system, and behavior patterns, such information should be carefully considered. What we require therefore are statistically meaningful ways of incorporating existing data into the modeling and decision-making process to improve the reliability of long-term forecasts.

PVA's proponents sometimes give the impression that it is a panacea; while it's detractors imply that it is futile. It is neither. It is a tool that through practice and refinement can become increasingly useful. It is just important that practice and theory continue to develop together.

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Book Review

Landscape Ecology and Resource Management: Linking Theory with Practice

John A. Bissonette & Ilse Storch, eds.
Island Press, 2003

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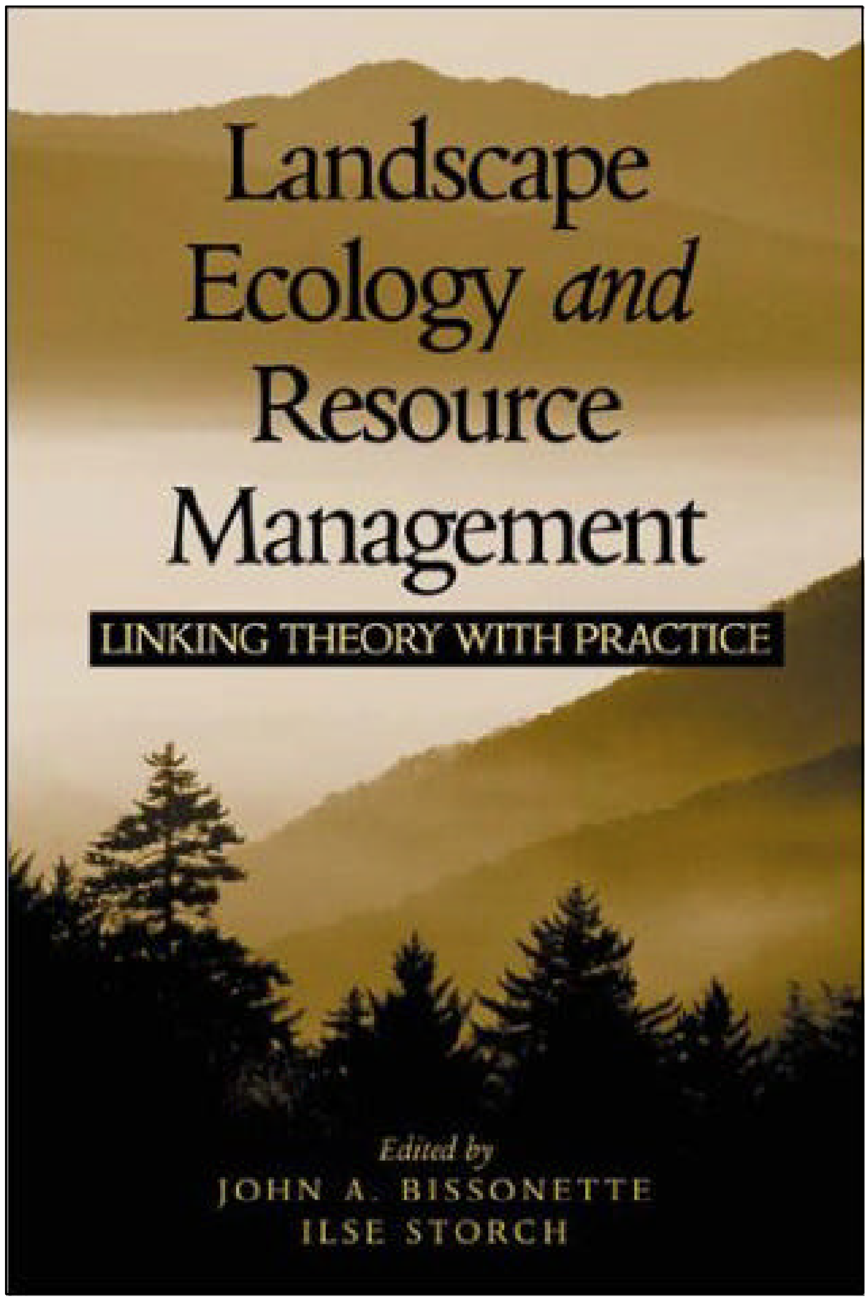

Researchers and managers working to protect rare species are increasingly becoming aware that the spatial pattern of critical habitats and temporal variation in ecological conditions from disturbance or other processes can have strong effects on focal species' population dynamics. As the "landscape" branch of ecology has grown, many researchers have suggested that this field's theoretical focus on understanding how spatial and temporal heterogeneity influence species could provide many insights to those working specifically on maintaining viable populations of threatened or endangered species over the long term. Although recent shifts toward larger-scale planning and an appreciation of the importance of the landscape "matrix" (the form of land use or habitat that surrounds a focal patch of habitat), suggest that theories from landscape ecology are influencing management activities, the rate of information transfer has clearly lagged behind recent theoretical developments in the field. This lack of communication is harmful in both directions, as the extent to which conservation efforts are designed without incorporation of potentially useful landscape concepts slows progress in the theory-development side of this branch of ecology due to a lack of field-based evaluations of new ideas. A new edited volume, *Landscape Ecology and Resource Management: Linking Theory and Practice,* by John A. Bissonette and Ilse Storch (2003, Island Press) was designed to help bridge this communication gap between landscape ecologists and conservation practitioners.

The specific goal of *Landscape Ecology and Resource Management: Linking Theory and Practice* is to help those involved with conservation "on the ground" become familiar with and implement concepts and analytical tools from landscape ecology. The first 12 chapters serve as tutorials on particular topics or tools, and the last five chapters comprise a set of case studies. Bissonette and Storch have organized the tutorial section (chapters 1-12) into two parts; the first focuses primarily on tools for analyzing and quantifying landscapes, and the second examines how human actions and values can be integrated with landscape level information.

The first seven chapters of the "tutorial" section address the "Conceptual and Quantitative Linkages" between theoretical principles in landscape ecology, and management challenges faced by practitioners. Here, researchers from Europe and the U.S. present diverse topics in landscape ecology, many of which are likely to be of interest to those working to protect rare species. In particular, chapters on identifying and interpreting spatial patterns of species distributions on the landscape, and the use of fitness landscapes as a tool for predicting habitat use are likely to provide thought-provoking reading for many managers. Throughout these seven chapters, the authors reinforce the importance of explicitly considering both the scale of measurement and data analysis, and the role of spatial and temporal heterogeneity. This section also introduces readers to more specific concepts and approaches such as the potential for thresholds in landscape structure, and many forms of spatial modeling, all within the context of addressing resource management problems. In particular, practitioners working with rare species are likely to appreciate a review of empirical studies testing how well landscape theories can help predict variation in vital rates and an example of how vegetation and wildlife models utilizing data collected at different scales can be merged for the purpose of predicting suitable habitat.

In the second half of part one (tutorial chapters), "Linking People, Land Use, and Landscape Values", the focus of the chapters shifts from ecological theories and pattern analysis to the importance of considering the role human activities and societal values play in management. These five chapters explicitly integrate human land uses (primarily agriculture, hunting, grazing, and forestry) with landscape processes and patterns, and focus on topics ranging from using management to mimic forest disturbance regimes, a "neuro-fuzzy" habitat model for exploring potential changes in agriculture on target species and challenges to conserving large mammal populations in an Ugandan park. This section contrasts three chapters describing European sites with very long histories of intensive management with challenges facing regional planners in northern Australia and National Park managers in Uganda. For those primarily interested in the human dimension of resource management challenges, I felt these two chapters describing work in the more "natural" areas of northern Australia and Uganda were most effective at conveying the importance of understanding the culture and values of local people in crafting management strategies. Many North Americans are likely to find Almo Farina's emphasis on protecting "cultural landscapes" intriguing; in the systems he describes, the long history of land use in Europe has produced heterogeneous landscapes in which species richness is currently enhanced, rather than depleted, by active management.

The final part, "Linking Theory and Application: Case Studies" provides five in-depth examples of potential tests of the value of incorporating landscape ecology into wildlife management. The key word here is "potential", as these case studies are primarily providing the information (e.g., ecological data and models, cultural history) that would set the stage for implementing a management plan or set of conservation priorities, but are describing cases in which implementation is in progress, or has not yet been attempted. Hopefully this book will help promote the kind of work that will allow future volumes to evaluate case studies in a wider range of stages of implementation so that more information will be available to help practitioners focus in on the most useful tools and concepts to adopt from this diverse field.

A key strength of this book is that Bissonette and Storch have done an admirable job of collecting chapters that represent many geographic regions and approaches to landscape ecology, providing a very broad view of the range of ideas pursued by researchers in this field. Some readers might miss an introductory chapter or two on the "basics" of landscape ecology, however I found the integrated nature of presenting concepts along with relevant examples and potential applications to be very effective. (Most readers involved in endangered species work can probably skip the introductory chapter, as it primarily describes the "biodiversity crisis" as a motivation for incorporating more science into management). An additional strength of this book is that, although the authors clearly have high hopes for applying theories and tools of landscape ecology to conservation problems, the book as a whole presents a balanced picture of this young scientific field. The authors, especially in the first section, have identified areas both of great promise and areas where empirical data do not support current theories or where analysis tools run the risk of becoming more sophisticated without enough reality checks on whether the patterns identified are meaningful. For example, in Chapter 1, Bissonette makes a point of calling in to question the idea that all observed spatial patterns of organisms are necessarily relevant to managers, emphasizing that it is quite possible to detect different patterns for the same species when you examine patterns from different observational scales. By taking this balanced approach, the editors have produced a book with great potential to help facilitate a dialogue between practitioners and researchers that should help to accelerate progress from both the theoretical and applied branches of landscape ecology.

Incorporating Multiple Criteria into the Design of Conservation Area Networks

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Abstract

A two-stage protocol for the design of conservation area networks which allows multiple constraint synchronization is described. During the first stage areas are selected to represent components of biodiversity up to specified targets as economically as possible. The principal heuristic used is complementarity. This process results in a set of conservation area networks which comprise the feasible alternatives for the subsequent analysis. During the second stage, multiple criteria (including spatial configuration criteria, vulnerability criteria, and socio-political criteria) are used, first to select the non-dominated feasible alternatives, and then to refine the non-dominated set further. This refinement is performed using a modification of the analytic hierarchy process.

Resumen

Describimos un protocolo de dos etapas para el diseño de una red de zonas que permita la sincronización de múltiples limitantes. En la primera etapa, se eligen zonas representativas de la biodiversidad hasta obtener en la manera más económica posible las metas especificadas. La complementación es la heurística usada. Este proceso genera una red de áreas de conservación que constituyen en alternativas viables para ser analizadas subsecuentemente. Durante la segunda etapa, usamos criterios múltiple (incluyendo criterios de configuración espacial, de vulnerabilidad, y político-social) primero para seleccionar alternativas viables no dominantes, y luego para refinar aun mas la selección del grupo no dominante. Para lograr la selección usamos una variación del proceso de jerarquía analítica.

Introduction

Conservation areas consist of sites at which biodiversity management plans are implemented (Sarkar 2003). Traditional conservation areas include national parks and wildlife reserves; more recent categories include biosphere reserves and community conservancies. The first stage in the design of a conservation area network (CAN) consists of ensuring the adequate representation of all surrogates for biodiversity (for instance, species, ecosystems, habitats, *etc.*) in a network of selected places. Adequacy of representation is measured by the satisfaction of an explicit quantitative target of representation for each surrogate, such as, 100% of occurrences for a critically endangered species, or 10% of occurrences for a common species. In principle, these targets are supposed to reflect the biological requirements for the indefinite persistence of each surrogate. In practice, these targets often only reflect socio-economic constraints and are established by planners, usually in consultation with scientists (Soulé and Sanjayan 1998). Since not all areas of biological interest can be set aside for conservation because of competing claims on land, it is often imperative that this representation be achieved as economically as possible, with as few sites as possible being set aside for conservation (Margules *et al.* 1988).

The representation problem comes in two versions: 1) achieve the specified targets of representation for biodiversity surrogates in as few sites as possible, and 2), given a maximum budget of sites that can be included in a CAN by satisfying the targets of representation for as many surrogates as possible (Sarkar *et al.* 2004b). Both of these problems can be formulated as constrained optimization problems in the formalism of mathematical programming, and solved using "branch-and-bound" algorithms, which are guaranteed to produce the best solutions (Nemhauser and Wolsey 1988). However, these optimal algorithms are computationally inefficient and cumbersome to use. Consequently, conservation biologists have devised a variety of heuristic algorithms which solve the problems rapidly and generally achieve almost as much economy as the optimal algorithms (Csuti *et al.* 1997; Pressey *et al.* 1997). Most of these algorithms are based on the principle of complementarity (Margules *et al*. 1988; Justus and Sarkar 2002): sites are added iteratively to a CAN on the basis of how much representation they provide for surrogates which have not yet met their targets in the sites that are already selected. Other iterative heuristic rules that have been commonly used include the prioritization of sites by the rarity of the surrogates present in them.

The second stage of network design is the refinement of the set of CANs which satisfy the biodiversity representation targets in order to incorporate other criteria. These criteria generally fall under three categories: 1) spatial configuration criteria (such as, size, connectivity, and dispersion of the conservation areas.); 2) persistence criteria (such as population viabilities, measures of threat and vulnerability); and 3) socio-political criteria (such as economic and political costs). These criteria are not mutually exclusive. For instance some spatial configuration criteria, such as size and connectivity, are usually also persistence criteria.

Refinement using these criteria is often difficult because of two reasons: 1) not all of the criteria can be directly measured on the same quantitative scale; and 2) typically, not all of them can be optimized simultaneously, requiring the use of trade-offs between the alternatives. Methods for the incorporation of such criteria into CAN design are currently a topic of ongoing research. (In some protocols for CAN design, some of these criteria are incorporated into the basic site prioritization process)

We describe here a two-stage protocol for CAN design and illustrate its use by analyzing a data set from continental Ecuador. This protocol uses a modified version of the analytic hierarchy process (AHP) to avoid some well-known paradoxes of the original version while maintaining consistency with traditional multiple attribute utility theory (MAUT). It should be stressed that the results presented here are intended only as an illustration of these methods; they are not intended to guide policy choices in the field without further refinement. We will then describe how the data set from Ecuador must be treated for use in a planning protocol. Subsequent sec-

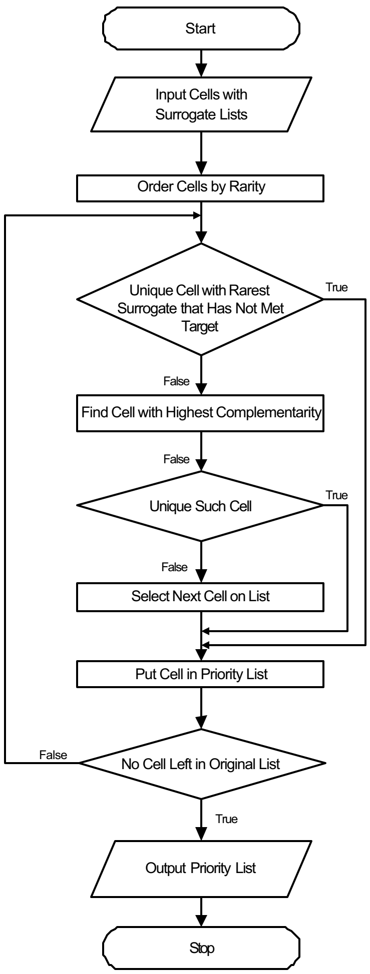

Figure 1. Rarity-Complementarity Algorithm for Site Prioritization. This algorithm belongs to the family of algorithms originally introduced by Margules *et al*. (1988). A rarity-complementarity algorithm is used because it is generally known to give economical solutions (Csuti *et al.* 1997; Pressey *et al.* 1997). However, Sarkar *et al.* (2004b) have recently observed that pure complementarity algorithms also perform as well when probabilistic data are used. The algorithm used to generate the results used in the text differs from this basic procedure in three ways: (i) there is a test for adjacency after the test for complementarity; and (ii) the exit condition is the satisfaction of targets for all surrogates—see the text for more detail.

tions will then show how these data can be used for biodiversity representation and subsequent multicriteria analysis. The software necessary to use this protocol can be freely downloaded from the web.

Data Preparation

The type of data transformations that are required for systematic conservation planning will be illustrated using a data set for continental Ecuador (excluding the Galápagos Islands) which, with an area of 248 750 sq. km sq. km., is small in size but rich in biodiversity. Since geographical distributions of species are not currently available for a representative set of taxa, systematic conservation planning must be based either on abiotic environmental surrogates or modeled distributions of coarse biological surrogates. This analysis started with a 200 ´ 200m raster grid on which the modeled distributions of 46 major vegetation types were mapped. These vegetation types span the entire floral range of Ecuador. (See Sierra [1999] and Sierra *et al.* [2002] for details on the classification and modeling of the distribution of the vegetation types.) At this spatial scale, each data cell contains one vegetation type. This scale of resolution was reduced to a 2 ´ 2 km grid in which each new cell consisted of 100 of the original cells. The motivations for the scale change were to improve computational efficiency because of the reduced size of the data set and to use sites that are of appropriate size to be regarded as units of conservation.

The analysis kept track of the vegetation types in each of the original cells that were compounded to make a new cell. Each of the new cells can potentially contain at most 46 vegetation types. For each cell, for each vegetation type, the probabilistic expectation of the presence of that type in that cell was set equal to its proportion in the original 100 cells. Thus, if all the original cells contain exactly the same vegetation type, then that type has an expectation equal to 1 and each other type has an expectation of 0. Place prioritization algorithms have recently begun to use expectations because they can represent abundance data for surrogates (Sarkar *et al*. 2004b). Traditionally, these algorithms have only used data that are of surrogate presence (represented by 1) or absence (represented by 0).

The map of Ecuador was further modified by masking areas that were permanently transformed by anthropogenic modification as of 1996 (see Sierra *et al*. [2002]) and are, therefore, inappropriate for inclusion in a CAN. In this way 39% of the cells were excluded . The Ecuadorian national reserve system (NRS) was also represented on a 2 ´ 2km grid. The target of representation for each vegetation type was set to 10 % of the untransformed area in which that type occurred. Thirteen of the 46 vegetation types do not meet this target within the NRS. Any target of this type is a social choice reflecting a compromise between assessments of what is politically achievable and what is biologically desirable. The 10 % target is consistent with that proposed by the International Union for the Conservation of Nature (IUCN) (1994). A slightly higher target of 12% (though of the total land area and not for the habitat of each biodiversity surrogate) is currently being used for Canada (Hummel 1995), and much higher targets have occasionally been proposed (*e.g*., Ryti 1992). The protocol being discussed here can be carried out for any explicit target.

Site Prioritization

Given a list of cells (with each cell representing a site for potential inclusion in a CAN) and a list of the probabilistic expectations of biodiversity surrogates for each cell, a variety of algorithms can be used to select sites for inclusion in a CAN. The basic form of the algorithm used here is shown in Figure 1 (see also Sarkar *et al.* [2002]). Two additional steps were implemented. First, when ties remain after selecting cells on the basis of rarity and complementarity, cells that are adjacent to ones already selected are given preference. This preference for adjacency results in larger conservation areas. Second, the selection process terminated as soon as each vegetation type achieved its 10 % target of representation. The selection procedure was initiated using the existing NRS of Ecuador. Thus, the final solution records the minimum number of cells that must be added for the satisfaction of those targets according to this heuristic algorithm.

One-hundred different solutions were generated using randomized re-orderings of the data set. These re-orderings generated different solutions because they resulted in the selection of different cells when ties were broken by lexical order (that is, by selecting the next cell in the list of cells). All computations were carried out using the ResNet software package (Garson *et al*. 2002). Figures 2 and 3 show two of the solutions generated in this fashion.

In general, iterative procedures, such as the one used here, have the advantage that the biological reason for the selection of a cell in a CAN is explicitly known (for instance, whether it is selected because it contains more rare surrogates than other cells, has a higher complementarity value, or is adjacent to previously selected cells). Data of this sort facilitate the selection of alternative sites if, for unforeseen reasons, an initially selected site cannot be included in a CAN. However, less transparent procedures such as simulated annealing have also been successfully used for site prioritization (see Possingham *et al*. [2000]).

Multiple Criteria

Because each potential CAN obtained from the site prioritization stage satisfied the surrogate representation targets, from the perspective of biodiversity representation, each such CAN is an appropriate solution: these are called alternative "feasible" solutions. The second stage of CAN design consists of incorporating other criteria to rank the feasible alternatives. This stage is critical to conservation planning for two reasons: 1) selecting CANs is of practical value only if these are implemented as a part of a conservation plan. Implementation always occurs in socio-political contexts in which biodiversity conservation and other potential uses of the land (including agricultural development, industrial development, biological resource extraction, mineral resource extraction, recreation) must be negotiated; and 2) mere representation of biodiversity does not ensure its persistence into the future. The vulnerability of biodiversity components due to both biological and non-biological features must be taken into account.

The second stage consists of three steps: 1) an identification of the relevant criteria and the ranking of the solutions or "alternatives" according to each criterion; 2) the determination of a set of "non-dominated" (or "efficient") alternatives; and 3), if the non-dominated set is too large, further refinement of this set to find a final set of preferred alternatives. Sometimes step 3) is carried out for the entire set of feasible alternatives without first finding the non-dominated set. The entire second stage falls within the scope of multiple criteria decision making (MCDM) which consists of a variety of heuristic optimization methods as well as the well-developed multiple attribute utility theory (MAUT) and closely related variants such as the analytic hierarchy process (AHP) (Dyer 2004).

As noted earlier, the criteria to be incorporated fall into three categories which are not mutually exclusive: spatial configuration criteria; persistence criteria; and socio-political criteria. For the Ecuador data set, six criteria were used:

(1) *the aggregate number of conservation areas*, which should be minimized to achieve spatial cohesiveness of CANs;

(2) *the average area of each conservation area*, which should be maximized to encourage larger conservation areas. (This aspect of CAN design was also encouraged by the use of the heuristic rule preferring adjacency during the first stage);

(3) *the variance of the areas*, which should be minimized to discourage further the selection of very small areas;

(4) *the aggregate distance of the selected cells to existing units of the NRS*, which should be minimized, again to increase cohesiveness (the distances being calculated between the centroids of the nearest cells);

(5) *the aggregate distance to anthropologically transformed areas*, which should be maximized to decrease the threat of habitat destruction (the distances once again being calculated from the centroids of the nearest cells);

(6) *the total area of the selection cells*, which should be minimized to decrease the cost of acquisition of the added cells.

Criteria (1) –(4) are spatial configuration criteria; criterion (5) is a persistence criterion. However, both criteria [2] and [3] are also persistence criteria. Criterion (6) is

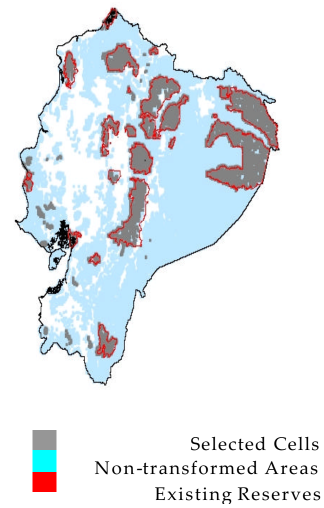

Figure 2. Best Solution for Ecuador Representing Biodiversity and Incorporating Six Additional Criteria. Because the runs were initialized with the cells belonging in the National Reserve System (NRS), the vast majority of the selected cells are within the NRS. Note that some cells within the NRS were anthropogenically transformed and were ignored by the selection procedure. The habitats that are most inadequately represented in the NRS are in the southwest of the country.

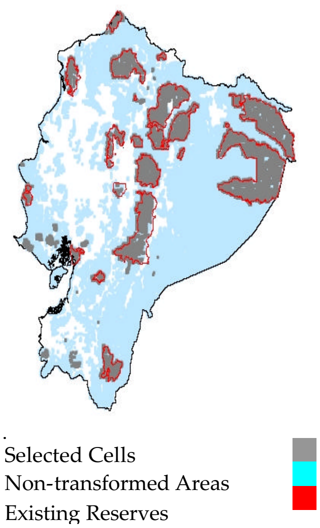

Figure 3. Second Best Solution for Ecuador Representing Biodiversity and Incorporating Six Additional Criteria. Because the runs were initialized with the cells belonging in the National Reserve System (NRS), the vast majority of the selected cells are within the NRS. Note that some cells within the NRS were anthropogenically transformed and were ignored by the selection procedure. The habitats that are most inadequately represented in the NRS are in the southwest of the country.

socio-political. In the protocol being described here it does not matter whether the criteria are independent of each other (Sarkar and Garson 2004). All 100 feasible alternatives were evaluated according to each of these criteria, which are such that a definite quantitative (numerical) value could be assigned to each alterative. For step 2 (though not for step 3 this is not essential: an ordinal ranking of each alternative according to each criterion is sufficient.

Turning to step 2, an alternative "dominates" another if: (a) it is no worse than the other according to any criterion; and (b) it is better than the other according to at least one criterion. A "non-dominated" alternative is one that is not dominated by any other alternative in the feasible set. Non-dominated alternatives correspond to the indifference curves of traditional economics. There is a natural sense in which non-dominated alternatives are special: each of these is an alternative that is uncontroversially better than all the dominated alternatives in the feasible set. Rothley (1999) advocated the use of non-dominated alternative sets in the selection of CANs; Sarkar and Garson (2004) provided a simple computationally efficient algorithm to find them. If the number of non-dominated alternatives is small, it makes sense to stop after finding them and turn over that set to political decision-makers who can then bring other non-modeled criteria to bear on them (Sarkar 2004). (Having more than one alternative enter the final political process of policy implementation is a virtue, not a limitation: it guards against the development of a biologically inferior plan should the plan originally proposed run into socio-political difficulties.)

Unfortunately, the number of non-dominated alternatives generally grows with the number of criteria. In practice, the non-dominated set must be further refined, which leads to step 3 of the second stage. For instance, in the case of Ecuador, using the six criteria listed above, 58 non-dominated alternatives were found which are clearly too many to be handed to political decision-makers in most contexts. (Figures 2 and 3 show two of these non-dominated solutions.) In step 3 each alternative must be numerically ranked according to each criterion, and the criteria themselves must be numerically ranked. However, the numerical ranking of the criteria are open to criticism as being arbitrary. This is why a CAN design process is usually regarded as more robust if it can stop at step 2 of the second stage (Sarkar 2004).

In standard MAUT a utility function is constructed to rank the non-dominated alternatives on the basis of their utility values (Keeney and Raiffa 1993; Dyer 2004). The AHP avoids the explicit construction of such a function. Instead, it elicits values on the users' implicit preference function by requiring a numerical pairwise comparison of the criteria on an increasing ratio scale, usually from 1 to 9 (Saaty 1980). This approach then generates weights, or scaling constants, for the criteria using the pairwise binary comparisons. A value of 1 indicates that the two criteria being compared have the same rank; a value of 9 indicates that changes over the range of values for the second is maximally preferred to changes over the range of values for the first. Thus, if criterion (*A*) has a ratio scale value of *X* compared to criterion (*B*), then criterion (*B*) has a ratio scale value of 1/X compared to criterion (*B*).

For the Ecuador data, the ratio scale ranking of the six criteria, taken in order, can be represented by the following matrix:

$$\begin{pmatrix} 1 & \frac{1}{2} & 9 & 3 & 6 & 7 \\ 2 & 1 & 9 & 9 & 4 & 9 \\ \frac{1}{9} & \frac{1}{9} & 1 & \frac{1}{5} & \frac{1}{6} & \frac{1}{2} \\ \frac{1}{3} & \frac{1}{9} & 5 & 1 & \frac{1}{3} & 4 \\ \frac{1}{6} & \frac{1}{4} & 6 & 3 & 1 & 2 \\ \frac{1}{7} & \frac{1}{9} & 2 & \frac{1}{4} & \frac{1}{2} & 1 \end{pmatrix}.$$

This means that changes over the range of values for criterion (2) was 1/2 as important as for criterion (1), while the changes for criterion (3) was 9 times as important as criterion (1), and so on. The eigenvector of this matrix with the highest eigenvalue provides the rankings of the criteria, which is essentially one approach to averaging the redundant comparisons. The rankings presented here were those that were found reasonable by one of the authors—they have no further claim of veridicality. The consistency of such elicited rankings can be checked, and the process iterated until an acceptable consistency level is found. The analysis of the Ecuador data set used the MultCSync software package to generate these rankings, test for consistency, and to support the subsequent analysis reported below.

The use of the AHP has been advocated in conservation planning many times (Anselin *et al.* 1989; Mendoza and Sprouse 1989; Kangas 1993; Peterson *et al.* 1994; Li *et al.* 1999; Mendoza and Prabhu 2000; Diaz-Balteiro and Romero 2001; Pesonen 2001; Reynolds 2001; Schmoldt and Peterson 2001; Clevenger *et al.* 2002; Villa *et al.* 2002; Ananda and Herath 2003), though, previously, only over the entire feasible set, without its initial refinement to a non-dominated set. Moreover, because the original AHP compounds the ranking of preferences and of the alternatives after normalizing both sets independently, this strategy leads to the paradox of rank reversal: the final ranking of two alternatives may change if new alternatives are added to the set (Belton and Gear 1982). Consequently, a modified algorithm, originally proposed by Dyer (1990), was used which avoids this problem. This modification is believed to help ensure consistency between the AHP and traditional MAUT (Kamenetzky 1982; Belton 1986; Dyer 1990; Salo and Hämäläinen 1997).

The two alternatives shown in Figures 2 and 3 are the two best alternatives found in this way, taking all six criteria into account. They select different areas in southwestern Ecuador thus potentially offering a range of alternative choices to political decision-makers. Since all non-dominated alternatives are ranked, a set of best alternatives can be presented to such decision-makers, with the number of alternatives to be presented determined by the decision-making context.

Final Remarks

The protocol described here is not the only option for incorporating multiple criteria into CAN design. An alternative strategy is to incorporate these criteria at the iterative step of selecting individual cells for inclusion of a CAN. Faith and Walker (1996) have developed such a protocol, based on complementarity, though only for two criteria (biodiversity representation and cost). Possingham et al. (1990) have developed a different such protocol, based on a simulated annealing algorithm (Kirkpatrick *et al.* 1983), but only for three criteria (biodiversity representation, area, and shaper). The main difference between the "global" strategy of the protocol described here and such a "local" strategy is that the former privileges biodiversity in the sense that every feasible alternative incorporates the representation of all biodiversity surrogates up to the specified target. In contrast, in the local strategy, some biodiversity surrogates may not achieve their target.

Acknowledgments

We thank Jim Dyer for helpful conversations and comments on an earlier version of this paper.

Software Availability

The initial prioritization of sites was carried out using the ResNet 1.2 software package (Garson *et al.* 2002). Multiple criterion synchronization used the MultCSync 1.0 software package (Sarkar *et al.* 2004a). Both software packages can be freely downloaded from http://uts.cc.utexas.edu/~consbio/Cons/ResNet.html.

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Monitoring the state-endangered Common Raven (*Corvus corax*) in southeastern Kentucky

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Abstract

In the eastern U.S., direct killing by humans and the loss of forest habitat and large mammals during the late 19th and early 20th centuries caused the extirpation of ravens in all but the most inaccessible and rugged portions of the Appalachian Mountains. Remnant raven populations in these areas have since served as a source of individuals that have successfully colonized portions of its former range. However, in many areas that appear suitable for its recovery, such as the Cumberland Plateau of Kentucky, the raven has yet to reestablish. We speculate as to what factors may be responsible for the failure of ravens to repopulate eastern Kentucky and outline our plan to assess its status in this region.

Resumen

En el este de los Estados Unidos, la perdida de bosque de hábitat y las matanzas de mamíferos grandes durante los siglos 19 y 20 causaron el exterminio de cuervos en todas las regiones de la cordillera Apalachian excepto las zonas menos accesibles. Las poblaciones remantes de cuervos han servido como una fuente de individuos que han colonizado porciones de su dominio anterior. Sin embargo, en áreas que parecen ideales para su recuperación, como la planicie de Cumberland, el cuervo no ha logrado reestablecerse. Especulamos sobre las razones del fracaso de los cuervos de volver a establecerse en el oriente de Kentucky y presentamos nuestro plan para analizar el estado actual de esta región.

The common raven, *Corvus corax,* is the largest-bodied passerine and one of the most globally widespread bird species (Boarman and Heinrich 1999). Although the raven once ranged throughout much of North America (Wilmore 1977), persecution by humans and the loss of forest habitat during the last two centuries reduced raven abundance and restricted its distribution to rugged and remote portions of its range (Wilmore 1977). In the eastern United States, the more inaccessible portions of the Appalachian Mountains served as the last stronghold of the raven during the early twentieth century. Raven Rock, Raven's Window, and Raven Gap are just a few of the dozens of place names attached to natural features throughout the eastern U.S. that commemorate the former more widespread occurrence of its animal namesake.

The common raven occurred throughout Kentucky during early European settlement, but it was most notably abundant in the mixed-mesophytic forests of the southeastern Cumberland Mountains and Cliff Section of the Cumberland Plateau (Mengel 1965; Palmer-Ball 1996). Although Mengel (1965) suggested that the raven was extirpated from Kentucky by the late 1950's, ravens have been occasionally observed in several locations in southeastern Kentucky since 1970 (Croft 1970; Davis et al. 1980; Heilbrun 1983; Smith and Davis 1979; Stamm 1981). However, it wasn't until the mid-1980s that ravens were found to nest in this region (Fowler et al. 1985). More recently and nearly 50 km northwest, ravens have been observed nesting in cliffs created by surface mining (Larkin et al. 1999; Cox et al. 2003), a phenomenon also documented in Pennsylvania (Brauning 1992).

Elsewhere in North America during the past two decades, the raven has recolonized portions of its former range (Kilham 1989; Saemann 1989) and increased in abundance (Buckelew and Hall 1994; Boarman and Berry 1995). Raven recovery has been attributed to factors that include an increase in older forests, behavioral adaptations to human landscapes, and increases in large herbivore populations that have provided more road-killed carrion (Buckelew and Hall 1994; Boarman and Heinrich 1999).

Although source populations exist in Tennessee, Virginia, and West Virginia (Buckelew and Hall 1994; Nicholson 1997), the raven has yet to recolonize greater than 95% of eastern Kentucky even though they are capable of dispersing hundreds of kilometers (Heinrich 2000). Moreover, limited recolonization has occurred despite regional trends that should favor its recovery such as increased forest cover, exponentially higher numbers of white-tailed deer, *Odocoileus virginianus*, and resultant roadkill than in past decades, and a regional decline in human population. Areas such as the Red River Gorge, Breaks Interstate Park, Big South Fork, and other cliffy sites in the Cumberland Plateau appear to have an abundance of suitable nesting sites for the raven. Further, a growing reintroduced elk, *Cervus elaphus,* population that likely exceeds 3000, as well as an established coyote population in southeastern Kentucky (Cox 2003) have provided the raven with an additional food resource and the means by which to exploit it, respectively. Unlike portions of the western U.S., predation does not cause significant mortality of elk in Kentucky (Larkin 2001; Cox 2003; Seward 2003). However, elk that succumb to meningeal worm infection, are killed by automobiles, or harvested by hunters should provide a consistent year-round source of carrion for the raven. In fact, ravens at one locale have been observed scavenging on both elk (Cox et al. 2003) and white-tailed deer (A. Miller, pers. comm., University of Kentucky). These facts suggest that the availability of nesting sites and food are not limiting factors to raven recolonization of the region.

Sensitivity to human disturbance and low survival of fledglings are two other factors that may be impeding raven recovery in the eastern U.S. Although the raven has adapted to and often thrives in human disturbed areas in the western U.S. (Boarman and Berry 1995, Knight et al. 1993, White and Tanner-White 1988), its eastern counterpart may be less tolerant of human activity. This could explain why the raven inhabits rugged, high elevation areas and its reluctance to expand into or inability to survive in what appears to be otherwise suitable low-elevation habitat.

Research on the raven in the central and southern Appalachians, however, is scant due to the difficulty of locating and monitoring the species in rugged, relatively roadless terrain. In order to obtain a baseline estimate of abundance and distribution of the common raven in southeastern Kentucky, we plan to conduct a multiyear survey of the most rugged and roadless areas in this region. Our survey will include Cumberland, Pine, and Black Mountains that are parallel to and traverse most of the length of the Kentucky-Virginia border. We will conduct visual count surveys (Marquiss et al. 1978) as well as playback calls to elicit vocal responses from ravens that may occupy these areas. Because ravens frequent and often benefit from human-altered landscapes in the western U.S. (Boarman 1993), we also plan to survey active landfills in this area to determine if they are being used by the species. Once our survey is completed, we will evaluate the potential for future research intended to characterize demography, habitat use patterns, attributes of active nest sites, and food habits. Future studies should also examine fledgling survival and dispersal patterns to determine the fate of those individuals born in Kentucky.

We hope that our research will at minimum be able to begin to document population trends of ravens within the region and to better understand what factors allow this elusive corvid to persist in this region of the state and not others. These research efforts will assist wildlife agencies in identifying suitable habitat appropriate for reintroduction of ravens if repatriation of the species to portions of its former range becomes a management priority. Finally, given the continued large-scale surface mining and recent increase in logging in eastern Kentucky (Kentucky Environmental Quality Commission 2001), our efforts may provide insight into how such extractive activities affect the ecology and recovery of the common raven.

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