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Contents

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Endangered Species UPDATE
School of Natural Resources and Environment
The University of Michigan
440 Church Street
Ann Arbor, MI 48109-1041
(734) 763-3243; fax (734) 936-2195
E-mail: esupdate@umich.edu
http://www.umich.edu/~esupdate

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The views expressed in the Endangered Species UPDATE

Using Expert-Opinion Surveys and GIS to Model Potential Cougar Habitat and Dispersal Corridors in Midwestern North America

Michelle A. Larue
Clayton K. Neilsen

Cooperative Wildlife Research Laboratory
Southern Illinois University
Carbondale, IL 62901

Abstract

Confirmations of cougar (*Puma concolor*) presence in Midwestern North American have increased considerably during the last decade. Although increasing cougar presence in the region may be indicative of potential eastward expansion of current cougar range via dispersal, no research has been conducted on cougar potential in the Midwest. Herein, we describe our approach to modeling potential cougar habitat and dispersal corridors in the Midwest (i.e., nine states and two provinces) using expert-opinion surveys, geospatial data, and a geographic information system (GIS). We intend to identify the distribution of potentially suitable habitat in this region where empirical data regarding cougar habitat use is not available. We will also use the map of potential habitat suitability, expert knowledge, and a GIS to evaluate potential dispersal corridors for cougars. Our results will provide information to wildlife biologists for management support, protection, and public education regarding cougar presence in the Midwest.

About the Authors

Michelle A. LaRue is a graduate research assistant at the Cooperative Wildlife Research Laboratory at Southern Illinois University, Carbondale. She is interested in conservation biology, particularly in using modeling techniques to better understand habitat suitability and effects of landscape changes on mammalian populations.

Clayton K. Nielsen is a Wildlife Ecologist at the Cooperative Wildlife Research Laboratory at Southern Illinois University, Carbondale. He is a Certified Wildlife Biologist, President of the Illinois State Chapter of The Wildlife Society, and Director of Scientific Research for the Cougar Network. Clay and his graduate students study mammalian ecology and management in a variety of landscapes

Abstract
Confirmations of cougar (*Puma concolor*) presence in Midwestern North American have increased considerably during the last decade. Although increasing cougar presence in the region may be indicative of potential eastward expansion of current cougar range via dispersal, no research has been conducted on cougar potential in the Midwest. Herein, we describe our approach to modeling potential cougar habitat and dispersal corridors in the Midwest (i.e., nine states and two provinces) using expert-opinion surveys, geospatial data, and a geographic information system (GIS). We intend to identify the distribution of potentially suitable habitat in this region where empirical data regarding cougar habitat use is not available. We will also use the map of potential habitat suitability, expert knowledge, and a GIS to evaluate potential dispersal corridors for cougars. Our results will provide information to wildlife biologists to support management, protection, and public education regarding cougar presence in the Midwest.

Increasing Cougar Presence in Midwestern North America

Historically, cougars (*Puma concolor*) occupied most of the western hemisphere, ranging from the Atlantic to Pacific oceans and from northern British Colombia to southern Chile (Sunquist and Sunquist 2002). However, by the late 1890s these top predators were extirpated from eastern North America because of habitat loss and intentional killing due to concerns about human safety, game populations, and livestock depredation (Sunquist and Sunquist 2002). Populations of cougars within North America have since been restricted to the West, with the exception of the small Florida panther (*P. c. coryi*) population in southern Florida. Currently, cougars are found in only one-third of their historical range in North America (Pierce and Bleich 2003), although cougar distribution throughout the Western Hemisphere is still the largest of any terrestrial mammal (Sunquist and Sunquist 2002).

Recently cougars have surfaced as a topic of discussion among wildlife biologists and the general public due to the possibility of dispersal and natural re-colonization east of their current geographic range. Although sightings of cougars may be unreliable, confirmed cougar carcasses, scat, and tracks (i.e., cougar "confirmations"; Figure 1) in Midwestern North America (hereafter the Midwest) have increased dramatically during the past 15 years suggesting eastward movement of cougars (Nielsen et al. 2006). For example, the Cougar Network reports >120 cougar confirmations since 1990; in Nebraska alone there have been 24 cougar confirmations during this period (Cougar Network 2006). Furthermore, Iowa and Missouri combined report 15 cougar confirmations since 1990 (Cougar Network 2006).

Many cougar confirmations exist as carcasses of young males, which are the primary dispersers in cougar populations (Sweanor et al. 2000, Logan and Sweanor 2001). Recent research has found that cougars can disperse considerable distances, as evidenced by a juvenile male dispersing 1,067 km into Oklahoma from the Black Hills (Thompson and Jenks 2005) and a juvenile female dispersing 1,336 km within western cougar range (Cougar Network 2006). Given the increasing number of cougar confirmations and their long-distance dispersal capability (Sweanor et al. 2000), it is widely believed that cougars are attempting to re-colonize the Midwest via juvenile dispersal (Nielsen et al. 2006).

Although wildlife biologists require information to support management, protection, and public education regarding cougar presence in the Midwest, no information is available to assist such efforts. Herein, we describe our approach to provide the first information regarding cougar potential in the Midwest. Specifically, we are currently undertaking an effort to use expert surveys, geospatial data, and a GIS to model potential habitat and dispersal corridors for cougars in this region of profound human presence and landscape manipulation.

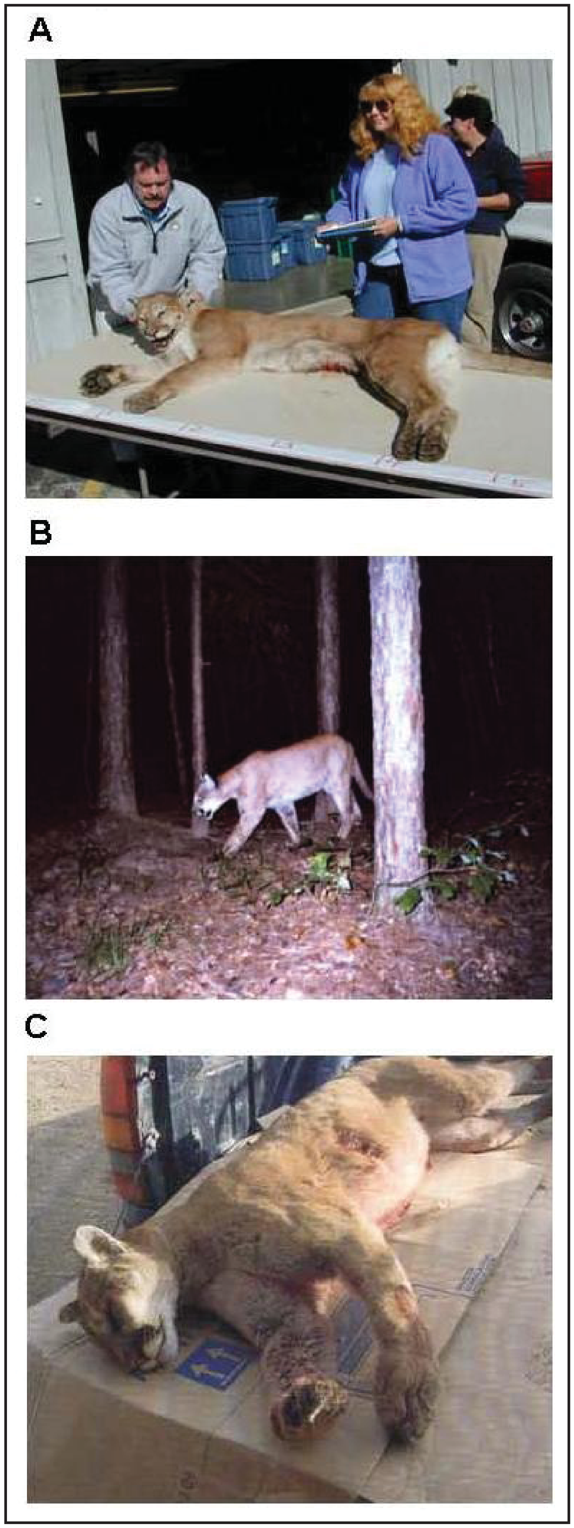

Figure 1. Example cougar confirmations as verified by the Cougar Network (2006). **A**: cougar involved in cougar-vehicle accident in Missouri, October 2002. **B:** cougar pictured by remote camera in Arkansas, August 2003. **C:** cougar shot by landowner in Manitoba, November 2004.

Approach to Modeling Potential Habitat and Corridors

Habitat models have been created for many carnivore species using animal location data, remotely-sensed land cover data, and multivariate statistics within a GIS (Clark et al. 1993, Carroll et al. 1999, Mace et al. 1999, Nielsen and Woolf 2002, Treves et al. 2004). Such analyses typically rely upon empirical data regarding species occurrence or habitat. However, these data are not available for cougar populations in the Midwest as they have been extirpated from the region for about 100 years (Sunquist and Sunquist 2002). To overcome this problem, we are using expert-opinion surveys (Store and Kangas 2001, Clevenger et al. 2002) in lieu of empirical data to provide information regarding potential cougar habitat requirements in the Midwest.

Our technique follows that of Store and Kangas (2001), where GIS, spatial analysis, and decision analysis techniques are used to develop large-scale habitat models. Our research is comprised of 2 primary objectives, to (1) develop a model of potential habitat, and (2) predict potential dispersal corridors. We will use expert opinion and multi-criteria evaluation, specifically the analytical hierarchy process (Saaty 1980), to transform expert knowledge regarding wildlife habitat needs into numerical form (Store and Kangas 2001). GIS applications will then be used to produce cartographic maps of cougar habitat by combining the expert-assisted data and spatial analysis of existing landscape

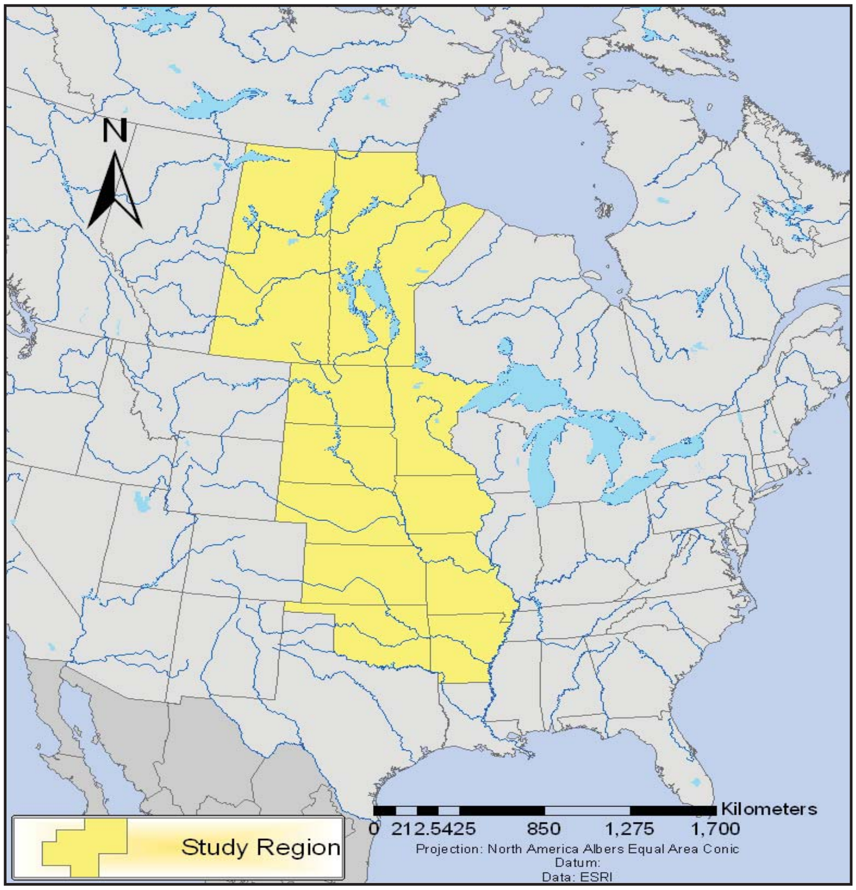

Figure 2. Study region for modeling potential cougar habitat and dispersal corridors.

information. These techniques have a history of use during the previous 5 years and are especially applicable in re-colonization analysis or reserve planning for rare species, such as Midwestern cougars. For example, Clevenger et al. (2002) combined and compared empirical data with literature and expert-assistance in the assessment of habitat linkages for grizzly bears (Ursus arctos), and reported that expert opinion closely reflected data gathered by radiotelemetry. Furthermore, Thatcher et al. (2006) used these methods to model potential reintroduction sites for Florida panthers.

We are modeling potential cougar habitat and dispersal corridors over a large portion of the Midwest, including the states of North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, Arkansas, Missouri, Iowa, Minnesota, and the provinces of Saskatchewan and Manitoba (Figure 2). These states and provinces were selected because of the number of cougar confirmations in the region, proximity to existing western cougar populations, and probability of suitable dispersal corridors, such as rivers (Nielsen et al. 2006).

Expert-Opinion Survey

In this modeling process, the first step in assessing potential habitat is to determine factors from existing studies and expert knowledge (Store and Kangas 2001). To obtain expert knowledge, we have used literature and expert assistance to develop a survey regarding potential habitat requirements of cougars in the Midwest. The survey consists of several questions regarding pair-wise comparisons of the following habitat factors: human density, distance to water, distance to roads, slope, and cover type. Prey (e.g., white-tailed deer, Odocoileus virginianus) densities will be assumed to be correlated with land cover because datasets are not available throughout the Midwest. Survey participants will be asked to score habitat variables in order of potential importance to Midwestern cougars, based upon personal experience and expert knowledge of cougar ecology. The survey will be sent to 25 western cougar biologists and furbearer biologists with knowledge of Midwestern landscapes.

Multi-Criteria Evaluation

The next step in the modeling process will be to produce map layers by transforming raw data into GIS layers (Store and Kangas 2001). This is carried out using a multi-criteria evaluation program outside of GIS (Store and Kangas 2001). Upon receipt of the completed surveys, we will evaluate the responses, which are contained in ranking matrices (Figure 3). We will begin evaluation by determining the relative importance of each habitat factor based on an optimization method where the importance of habitat factors is evaluated by pair-wise comparisons as applied in the analytical hierarchy process (Saaty 1980). Once each habitat factor has been assigned a relative priority, the modeling procedure will begin. This will consist of making the raw scores commensurable, weighting the standardized score maps, and then combining them into the model (Store and Kangas 2001).

Habitat Modeling

Cartographic modeling in ArcGIS 9 (ESRI 2004) will be performed to identify habitat potentially suitable for cougars in the region. Areas of suitable habitat will be identified by reclassifying and weighting each variable and subsequently mapping these variables and their associated weights by overlay analysis (Ormsby et al. 2004). This map will be produced using scores calculated and averaged in the multi-criteria evaluation, and will clearly classify areas of good versus poor potential habitat along a gradient of values.

Corridor Modeling

We will also conduct a GIS weighted-distance analysis and least-cost corridor analysis (Singleton et al. 2002) to map the effects of landscape barriers for cougars dispersing into the Midwest. We will map the linkages between habitat patches with fewest landscape barriers (Schippers et al. 1996). These analyses are complementary and are based upon the idea that movement can be mapped by assigning each cell within a map a relative "cost" of moving across the cell (Schippers et al. 1996, Singleton et al. 2002). This "cost" is calculated as the cell size times a weighting factor based on the habitat characteristics of the cell (Singleton et al. 2002).

The map of potential habitat will be the basis for the corridor model. A follow-up survey will be sent to the same wildlife biologists asking their expert opinion on corridor size and permeability of the landscape for cougars. A similar matrix will be created asking experts to rank the quality of habitat in regards to corridor movements, as well as identifying maximum distances cougars would likely travel within corridors.

Dispersal potential will be evaluated across the region based on road density, land cover, and distance to edge. The classes for each landscape variable will be given a value based on our experts' estimated resistance to movement (Singleton et al. 2002). Dispersal habitat suitability will then be calculated by multiplying each landscape characteristic value (Singleton 2002). This will result in an index of landscape permeability for cougars in the region.

A least-cost corridor analysis will then be conducted throughout the study region to identify the most permeable portions of the landscape for potential cougar dispersal. The same cost-weighting factors for the weighted-distance analysis will be used for least-cost corridor analysis, which will provide an index of overall difficulty of moving through an area (Schippers et al 1996, Wikramanayake et al. 2004). This will result in a map of likely pathways

Figure 3. Example of a ranking matrix from our expert survey used to develop models of potential habitat and dispersal corridors for cougars in Midwestern North America

	Barren/Developed and Open Water	Deciduous Forest	Evergreen Forest	Mixed Forest	Cultivated	Wetlands	Shrublands	Grasslands
Barren/Developed and Open Water	1							
Deciduous Forest		1						
Evergreen Forest			1					
Mixed Forest				1				
Cultivated					1			
Wetlands						1		
Shrublands							1	
Grasslands								1

that cougars could utilize for dispersal into the Midwest.

Importance of this Research

Although a few researchers have discussed the recent confirmations of cougars east of their range (Tischendorf 2003, Nielsen 2006, Nielsen et al. 2006), no studies have been conducted regarding cougar potential in the Midwest. The models and maps we produce will be important for several reasons. First, because of their role as a top predator, cougars will likely compete with other predators [e.g., grey wolves (Canis lupus), coyotes (Canis latrans), and bobcats (Lynx rufus)] in the Midwest (Pierce and Bleich 2003). This competition could result in character displacement or possible extirpation of other predator species. Cougar conflicts with wolves, for instance, may alter population characteristics, behaviors, and distribution of prey (Murphy et al. 1999). However, niche partitioning may also occur and allow for coexistence of cougars and other carnivores (Pierce and Bleich 2003). Regardless, our analyses will provide an important assessment of potential cougar distribution and where significant overlap with sympatric carnivores may occur.

Second, because white-tailed deer would be the primary prey species for cougars in the Midwest (Sunquist and Sunquist 2002), sports enthusiasts and wildlife conservation agencies are concerned about potential impacts of cougars on deer populations. White-tailed deer are the most important big game species in North America and their management injects millions of dollars into state wildlife conservation agencies and local economies (Miller et al. 2003). Given that competition with humans for game species (and concerns about cougar attacks on humans and livestock depredation, see below) was one of the primary reasons for cougar extirpation in the eastern portion of their range (Sunquist and Sunquist 2002), knowledge of potential distribution of cougars relative to white-tailed deer distribution is essential.

Third, concerns about potential cougar depredation of livestock (Torres et al. 1996) would need to be addressed through public education campaigns. The Midwest is an area of considerable cattle, swine, sheep, and horse production, and agriculturalists in the region are already worried about cougar depredation of livestock (C. Nielsen, personal communication). Finally, a serious implication of potential cougar re-colonization is the fear of cougar attacks on humans (Beier 1991, Kadesky et al. 1998, McKee 2003). An analysis of habitat potential could indicate where cougars may become established near centers of human populations or areas of livestock operations and prove an important educational and planning tool to address human-cougar conflicts.

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A Recovery Index:Developing a New Metric to Track Endangered Species Recovery Progress

Timothy D. Male[1,2]
Scott Walsh[1]
Michael J. Bean[1]

[1]Environmental Defense
1875 Connecticut Avenue NW
Washington, DC 20009 USA

[2] (202) 387-3500
fax: (202) 234-6049
tmale@ed.org

Abstract
The Endangered Species Act (ESA) faces continual criticism for having failed to achieve the recovery of more endangered and threatened species even though recovery is expected to be a long process for most species. However, little quantitative information is available to give a more nuanced picture of whether the ESA is succeeding in moving species toward recovery. In this article, we propose three methods of creating a 'Recovery Index' that uses annual population estimates for a subset of species to create an easily understandable index of how endangered species are doing from year to year. A Recovery Index based on annual population growth rates for 30 well-funded species showed a 320 percent increase in value over the period from 1985–2005. We believe that a Recovery Index like this, modified to include a more representative set of species, would be a useful new metric with which to track the success or failure of the Endangered Species Act.

About the Authors
Timothy Male (PhD., University of Hawaii) is senior ecologist at Environmental Defense where he works on USDA 'Farm Bill' legislation and the Endangered Species Act. His work is focused on demonstrating how national policy can be adjusted to better facilitate meaningful resource conservation on private lands.

Michael J. Bean (J.D., Yale University) is co-director of the Center for Conservation Incentives at Environmental Defense. He leads wildlife policy-making activities for Environmental Defense and oversees development and testing of incentive-based approaches to endangered species conservation, particularly on private lands.

Scott Walsh (MBA, University of Virginia) is a project manager in Environmental Defense's corporate partnerships program, which focuses on business approaches to environmental stewardship. He leads Environmental Defense's work with major companies to ensure the safe development of nanotechnology.

Introduction

In authorizing the Endangered Species Act of 1973 (ESA), Congress set forth the goal of protecting biodiversity by preventing species extinctions and promoting species recovery. However, the law continues be the subject of intense political debate in part because there are relatively few metrics by which to determine the law's success or failure in achieving this goal (USGAO 2006).

Required by the Government Performance Results Act of 1993 to quantify its performance, the U.S. Fish and Wildlife Service has set performance goals based on the number of species reported to be "improving" or "stable" in its own biennial reports to Congress (Male and Bean 2005). However, these categories are imprecise and often subjective. Moreover, inasmuch as the Fish and Wildlife Service makes these judgments, it is in effect grading its own performance. Performance measures that were more quantifiable and less prone to subjective judgment should provide a better measure of how much or how little progress is being made under the ESA.

Short of outright recovery and removal of species from the endangered species list, there are three overlapping criteria that are a component of most species recovery plans and by which recovery progress could be judged. These are: number of individual present in the wild, number of viable wild populations, and progress in removing or controlling the threats that endanger the species. What sort of metrics could be developed to track progress under any of these criteria?

The Dow Jones Industrial Average tracks the performance in stock price of 30 large U.S. companies that are meant to be representative of the U.S. market. The individual companies that comprise the Dow have multi-billion dollar revenues and collectively represent about one quarter of the value of the U.S. stock market. As an index, it is not the value of the Dow itself but the relative change in value over time that is useful in measuring 'blue chip' stock performance and as an indicator of the health of the U.S. economy. For example, from December 1973 when the ESA was signed into law until April 2006, the Dow grew by an annual compound growth rate of 8.3 percent.

We propose that a new index, a Recovery Index, be developed to track changes in the abundance of a subset of endangered species. Such an index would provide a useful indicator of the health of U.S. endangered species and give insights into some of the results of conservation efforts taken under the Endangered Species Act.

Methods

Just as a few dozen 'blue chip' companies represent a large share of the value of the U.S. stock market, there are a few dozen endangered species that receive the majority of the resources available for management, mitigation, and land acquisition. Approximately 85 percent of all federal and state expenditures on endangered species reported by the U.S. Fish and Wildlife Service in annual reports to Congress are spent on just 75 species.

Using U.S. Fish and Wildlife Service annual reports to Congress that document federal and state spending on each listed species, we ranked the 75 species that received the greatest total recovery and land acquisition funding by all agencies between 1989 and 2004, the period over which reports were available.

For these species, we searched the Internet for technical reports by federal or state agencies or peer-reviewed publications that reported on the total number of individuals of each species in any given year. We only included wild individuals in totals and most counts include only adults or breeding adults. We were able to find multiple years of range-wide census data from 1985

Table 1. Endangered and threatened species used to calculate Recovery Indices, including the estimated compound annual growth rate of wild populations of each species between 1985 and 2005.

Common Name	Scientific Name	Compound Annual Growth Rate
California condor	*Gymnogyps californianus*	0.1653
Least Bell's vireo	*Vireo bellii pusillus*	0.1650
Red wolf	*Canis rufus*	0.1601
Aleutian Canada goose	*Branta canadensis leucopareia*	0.1457
San Clemente loggerhead shrike	*Lanius ludovicianus mearnsi*	0.1249
Kemp's ridley sea turtle	*Lepidochelys kempii*	0.1154
California least tern	*Sterna antillarum browni*	0.0941
Southwestern willow flycatcher	*Empidonax traillii extimus*	0.0909
Bald eagle	*Haliaeetus leucocephalus*	0.0811
Interior least tern	*Sterna antillarum*	0.0804
Eastern brown pelican	*Pelecanus occidentalis*	0.0583
Florida panther	*Puma concolor coryi*	0.0582
West Indian manatee	*Trichechus manatus*	0.0557
Whooping crane	*Grus americana*	0.0539
American peregrine falcon	*Falco peregrinus anatum*	0.0539
Gray wolf	*Canis lupus*	0.0468
Key deer	*Odocoileus virginianus clavium*	0.0437
Coastal California gnatcatcher	*Polioptila californica californica*	0.0404
Atlantic right whale	*Balaena glacialis*	0.0343
Southern sea otter	*Enhydra lutris nereis*	0.0333
Red-cockaded woodpecker	*Picoides borealis*	0.0237
Piping plover	*Charadrius melodus*	0.0230
Hawaiian monk seal	*Monachus schauinslandi*	0.0224
Indiana bat	*Myotis sodalis*	-0.0107
Puerto Rican parrot	*Amazona vittata*	-0.0116
Wood stork	*Mycteria americana*	-0.0329
Cape Sable seaside sparrow	*Ammodramus maritimus mirabilis*	-0.0372
Steller sea-lion	*Eumetopias jubatus*	-0.0407
Atlantic salmon	*Salmo salar*	-0.1090
Razorback sucker	*Xyrauchen texanus*	-0.2511

to 2005 for 30 species. We chose these species because they are ones on which data were available to illustrate the way that such a Recovery Index could be developed, not because these are the ideal species upon which to base an index. Data on funding and population estimates, including the published sources for those estimates, is available from the first author on request.

In order to calculate an annual Recovery Index value we needed data from each species every year, but most species lacked such annual data. Therefore, we used the equation for compound annual growth rate to estimate population size in years with missing data. For species lacking population estimates from the last two years of the sampling period, we used growth rate to project population size based on previous years' data.

We used three methods to calculate Recovery Indices for these species. The Dow is calculated by summing the raw stock price (adjusted for dividend distributions) of all 30 component stocks and thus is weighted so that higher-priced stocks and those with greater variance over time have a greater influence on the index. Similarly, we calculated an abundance-weighted Recovery Index by simply summing up the total number of individuals for all species each year. For example, 1,409 Hawaiian monk seals (*Monachus schauinslandi*) would be added to the 19,142 wood storks (*Mycteria americana*) estimated to exist in 2001 and so on for the remaining 28 species.

Second, we calculated a standard-

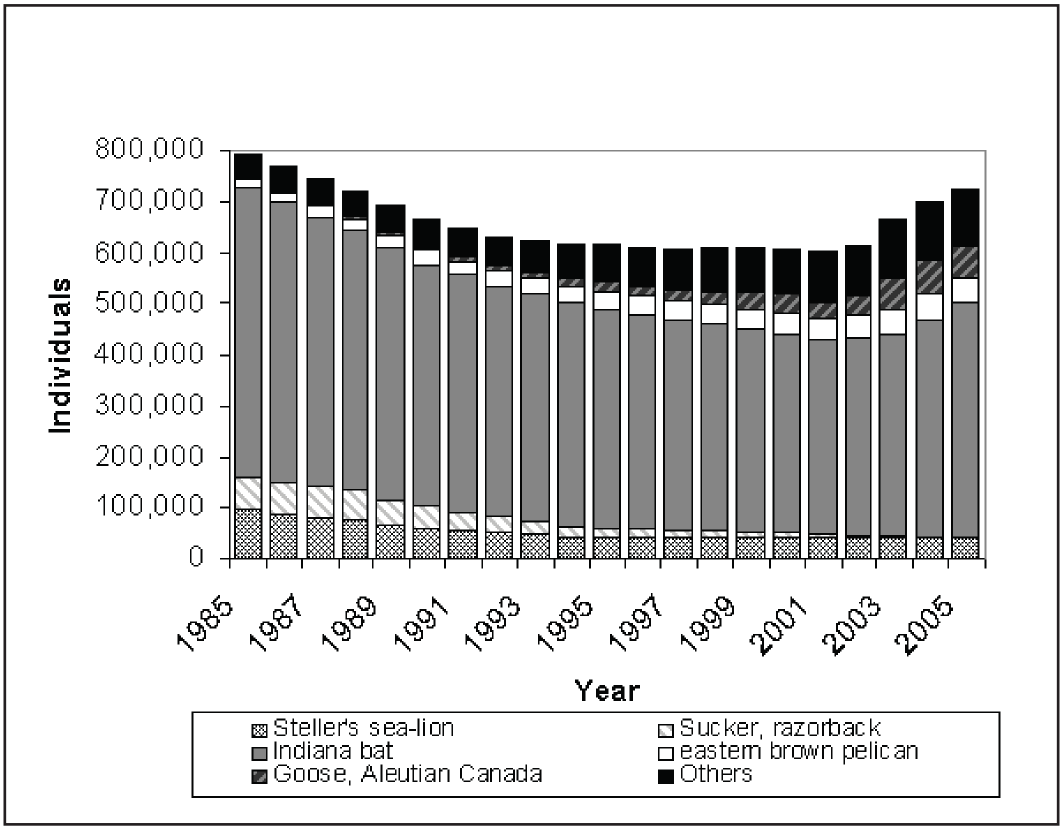

Figure 1. Abundance-weighted Recovery Index for 30 endangered species from 1985-2005, showing the overwhelming influence of the most numerous species on the overall value of the index.

ized Recovery Index that gave each species a similar influence on the value of the Index. We standardized each year of each species' population estimates by subtracting the mean population estimate for each species for all years and dividing by the standard deviation. For example, standardization changed the range of Indiana bat (*Myotis sodalist*) population sizes from 381,156 to 566,940 individuals to -1.25 to 2.19, while the Key deer population (*Odocoileus virginianus clavium*) went from 275 to 647 individuals to -1.17 to 1.13. Negative values represent population estimates less than the mean. The Recovery Index values were calculated by summing the standardized data for each species.

Finally, an increase in population size rather than absolute population size per se may be a better indicator of progress toward recovery, so we calculated a growth-weighted Recovery Index. This Index used the estimated percent change in species population sizes between one year and the next to calculate a mean growth rate across all species. For example, in 1990, the growth rate ranged from -46 percent for the Puerto Rican parrot (*Amazona vittata*) to +38 percent for the California least tern (*Sterna antillarum browni*) and a mean growth rate across all species of 3.1 percent. Negative values represent declines in estimated population size from one year to the next.

Results

Table 1 shows the species used to calculate indices and their estimated growth rate between 1985 and 2005. These 30 species represent less than one percent of listed species, but are the species on which Congress, the U.S. Fish and Wildlife Service, and other agencies have chosen to spend 25 percent of reported funding from 1989 to 2004. Across this time period, cumulative annual growth rate varied from -25.1 percent for the estimated wild adult population of razorback sucker (*Xyrauchen texanus*), an endangered Colorado River basin fish, to +16.5 percent for wild California condors (*Gymnogyps californianus*). The average growth rate across all species and years was +4.26 percent/year; seven species declined across the 20-year period from 1985 to 2005.

Abundance-weighted Recovery Index
The Recovery Index summing the total number of individuals of each species per year showed a -0.44 percent/year growth rate. Between 1985 and 2001, total abundance of these species declined by 1.71 percent/year, and thereafter the Recovery Index increased at an annual rate of 4.77 percent. This initial decline is driven by the influence of the large population size and 2.7 percent rate of decrease in the estimated Indiana bat population through 2001, a 5.1 percent annual decrease of surveyed Steller sea lions (*Eumetopias jubatus*) through 1994, and the decline of the razorback sucker, which continued throughout the period (Figure 1). Increasing populations of Aleutian Canada goose (*Branta canadensis leucopareia*) and eastern brown pelican (*Pelecanus occidentalis*) had a large influence over the Recovery Index after 1994. These two species were declared recovered and removed from the endangered species list during this time period.

Standardized Recovery Index
By first controlling for absolute abundance and variation in year-to-year abundance we produced a Recovery Index that gave each species similar influence over the rate of return on the Recovery Index, with no species influencing more than 4.1 percent of the variation in Index values. The compound annual growth rate for this index from 1985 to 2005 was 22.2 percent. The Index increased in every year during the sampling period except for a 56 percent decrease from 1989 to 1990 and a 4.1 percent decrease from 2001 to 2002. For stocks and mutual funds, analyses use hypothetical investments of $10,000 to allow comparison among investments. Similarly, after setting the Recovery Index at 10,000 in 1985, it had grown by more than 5,500 percent by 2005 (Figure 2).

Growth-weighted Recovery Index
Weighting the Recovery Index by growth rate produced an average annual growth rate across all years of 6.0 percent (figure 3). Growth rate for all species was negative only for 2001-2002. We calculated that if there were 10,000 hypothetical Recovery Index 'animals' in 1985, the population of those animals would have grown by almost 320 percent to 31,897 by 2005 (Figure 3).

Conclusion
The Recovery Indices modeled above provide new insights into progress in

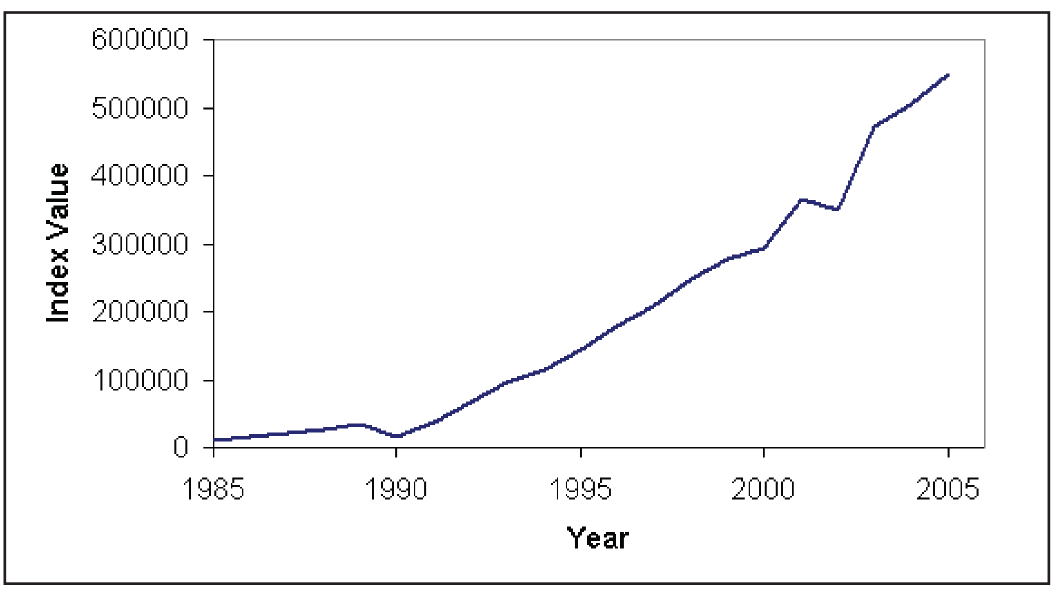

Figure 2. Standardized Recovery Index values from 1985-2005 showing the cumulative change in a hypothetical 10,000 individuals of the index 'species.' (Species population data was first standardized by subtracting the mean population size and dividing by the standard deviation within each species.)

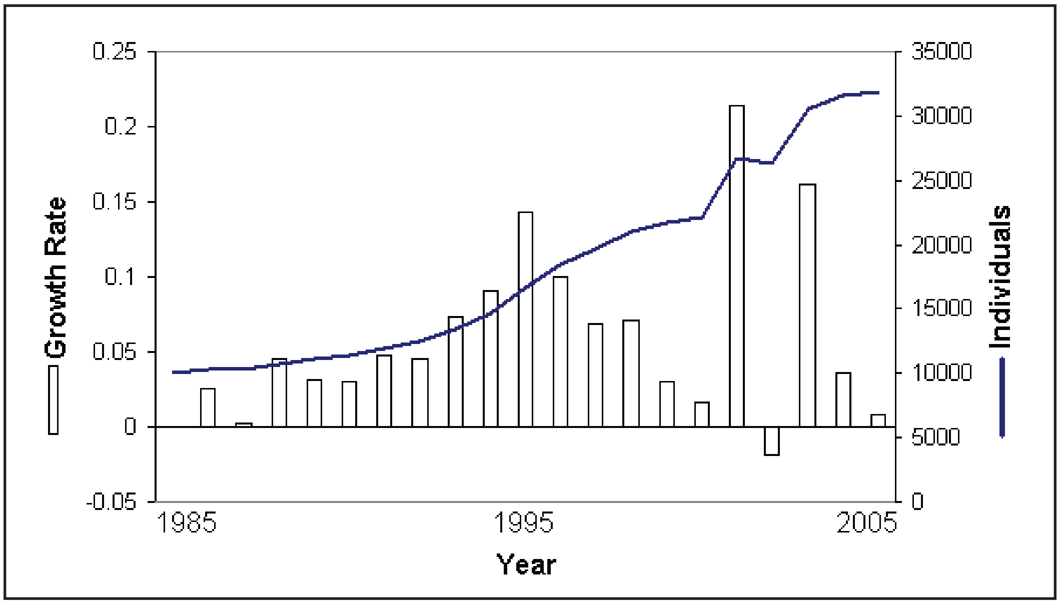

Figure 3. Growth-weighted Recovery Index from 1985 – 2005. Average annual growth rates for all species (bars; left y axis) and cumulative change in a hypothetical 10,000 individuals of the index 'species' based on these average annual growth rates (line; right y axis).

recovering species. Two of the three methods we used to calculate index values showed that numbers of these well-funded species have generally been increasing over a 20-year period. The third method showed increases since 2001. These indices are simple to calculate and are comprehensible by nonscientists in the same way that ups and downs of the Dow Jones Index or NASDAQ Composite are understood by millions of Americans.

The growth-weighted Recovery Index is likely to be the most useful metric to measure recovery progress for endangered species populations. The abundance-weighted Recovery Index is less robust because species varied dramatically in absolute population size and in variation among years, giving a handful of species a disproportionate influence on the index. For example, the 120,000 Steller sea lions estimated to exist in 1985 have a much more significant influence on a nonstandardized index than the 39 Puerto Rican parrots present in 1994. Overall, five species accounted for 78 percent of the variation in this index whereas no one species influenced more than 4.1 percent of variation in the standardized Recovery Index. However, the standardized Recovery Index is less useful because the mean and standard deviation for each species change with each new year of data, thus requiring the index to be recalculated for all years, every year.

We believe that it is reasonable to focus a Recovery Index on the best-funded species because these are the species upon which the government has chosen to focus the regulatory and incentive-based tools of the ESA. However, we make no claim that these 30 species are the appropriate species to include in any new Recovery Index. For example, birds, mammals, and fishes are disproportionately represented among the best-funded species and include a lower percentage of declining species than other taxonomic groupings (Male & Bean 2005). Before any further use of such an index, it would make sense to select a more taxonomically balanced group of well-funded species for which agencies have and will continue to conduct frequent range-wide population surveys. Species should also represent a diversity of life history traits (marine, aquatic, terrestrial, etc.) and regions of the country. It may also be inappropriate to use individual numbers for some species, such as annual plants that experience dramatic fluctuations in numbers between years. For such species, number of populations or element occurrences may provide a more robust dataset upon which to base any index (Wilcove et al. 2006).

A Recovery Index is only as good as the data upon which it is based and we have no doubt that changes in the methodology used to calculate population size, discovery of new populations, and chance variation in the percent of individuals censused all affected these indices by contributing to variation in estimated population size within species among years. Nevertheless, we feel that the changes in these indices reflect real change in the number of wild individuals of these species known to exist. Further, were agencies to decide to adopt such an index, doing so would create a strong incentive for them to collect increasingly accurate data in future years.

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Black Rhinoceros Conservation and Trophy Hunting in Southern Africa: Implications of Recent Policy Changes

Fred Nelson

School of Natural Resources and Environment
University of Michigan
Ann Arbor

fnelson@umich.edu

Abstract

The black rhinoceros, *Diceros bicornis*, is one of the most endangered species of terrestrial mammals in the world, with an estimated 3,600 animals remaining across a range that once covered most of sub-Saharan Africa. The principle cause of black rhino endangerment and decline during the past 30 years has been trade in rhino horn in the Far East and Arabian Peninsula. As a result of the threats posed from this trade, black rhinos have been listed on Appendix 1 of the Convention on International Trade in Endangered Species (CITES) since 1977, making all trade in rhino horn illegal. This trade ban has had limited impact in achieving its objectives of reducing the trade in rhino horn and protecting and recovering black rhino populations in Africa. Black rhino populations continued to decline in the late 1970's and 1980's, driven by the lucrative black market trade in horn and ineffective range state law enforcement practices. Several countries in southern Africa, principally South Africa, Namibia, and Zimbabwe, possess the most successful record of rhino conservation in sub-Saharan Africa; in South Africa and Namibia, black rhino populations have more than doubled since 1970. Rhino management in these countries has emphasized strong law enforcement and intensive monitoring in state protected areas, coupled with policies that enable private landholders and rural communities to capture economic benefits from rhinos. As a result of black rhino population recoveries in South Africa and Namibia, as well as the success of their market-based management strategies and desire to further expand black rhino ranges on private lands, those two countries submitted a proposal at the thirteenth CITES Conference of Parties (CoP) in October 2004, to initiate limited trophy hunting of black rhinos. Despite significant international resistance to rhino hunting among some conservation groups and animal welfare advocacy organizations, the proposal to hunt black rhinos in South Africa and Namibia was approved and quotas of five black rhinos per year for each of the two countries. This decision represents a watershed change in international approaches to black rhino conservation, with potentially important implications for the management and recovery of this critically endangered species.

About the Author

Fred Nelson is a Master's student in the School of Natural Resources and Environment at the University of Michigan. Prior to that he worked for seven years in Tanzania on community-based natural resource management, policy analysis, and enterprise development.

An Overview of African Rhinoceros Conservation: Regulations and Incentives

The black rhinoceros, *Diceros bicornis,* historically ranged across much of sub-Saharan Africa, particularly the savannahs and woodlands of the Sahel and east and southern Africa. The species was relatively common and widespread until the second half of the twentieth century, when hunting of black rhinos for their horn,(used in the Far East as a traditional medicine and to make ornamental dagger handles in Arabia) brought about rapid large scale reductions in range and numbers (Western 1987; Milliken et al. 1993; Emslie and Brooks 1999). The wild population of black rhinos declined from about 65,000 in 1970, to under 15,000 in 1980, and finally to a low of about 2,400 by the early 1990's (Figure 1) (Emslie and Brooks 1999). In Tanzania, for example, a population of almost 3,800 rhinos in 1980 was reduced to only 127 by 1992 as a result of rampant poaching (Figure 2) (Emslie and Brooks 1999).

With international trade in rhino horn driving these declines, the black rhino was transferred from Appendix II to Appendix I of CITES in 1977, making all trade in the species' horn and other products illegal. This strict trade ban did not, however, improve the species' conservation status. Rhino horn prices rose after the 1977 Appendix I listing, as traders stockpiled horn in response to the ban, and the continental black rhino population continued to plummet (t'Sas-Rolfe 2000). In general, CITES trade prohibitions have not been successful in reducing the demand for rhino horn and abetting the recovery of the species (Emslie and Brooks 1999; t'Sas-Rolfe 2000).

While the trade ban on rhino horn has been an ineffective basis for reversing the species' decline, several black rhino range states in southern Africa have recorded notable success in the face of widespread conservation failure. By the early 1990's, the few black rhinos that survived across the species' range resided primarily in heavily guarded and fenced government protected areas and private reserves. With the cost of protecting rhinos from poachers estimated at over $200 per km2/year (Leader-Williams 1990), these relatively small and fortified reserves were the only areas where rhinos could be effectively maintained. At the time, the majority of the remaining animals were in South Africa, Zimbabwe, and Namibia, with these three countries containing 77% of the continental population in 1990, an increase of 16% from the previous decade (Emslie and Brooks 1999). While the rest of Africa has lost nearly all of its black rhinos, populations in South Africa and Namibia increased between 1980 and 1997 from 630 to 1043, and 300 to 707, respectively (Figure 3) (Emslie and Brooks 1999). These two countries' black rhino populations have continued to increase during the past decade, to 1,286 in South Africa and 1,134 in Namibia at the time of the CITES conference in 2004 (CITES (nd)b; CITES (nd)c). South Africa and Namibia now contain about 70% of the estimated 3,600 black rhinos existing in Africa, making them the critical national actors in overall recovery efforts.

The success of these southern African nations in managing black rhinos has been a result of strong protected area management agencies, law enforcement, monitoring, and to a lesser degree the involvement of private landholders and rural communities. An important component of the overall wildlife management policies of Namibia and South Africa has been promoting locally managed commercial use of wildlife, and thereby encouraging the adoption of wildlife a form of private land use. Since the late 1960's, southern African countries have emphasized sustainable wildlife utilization, including commer-

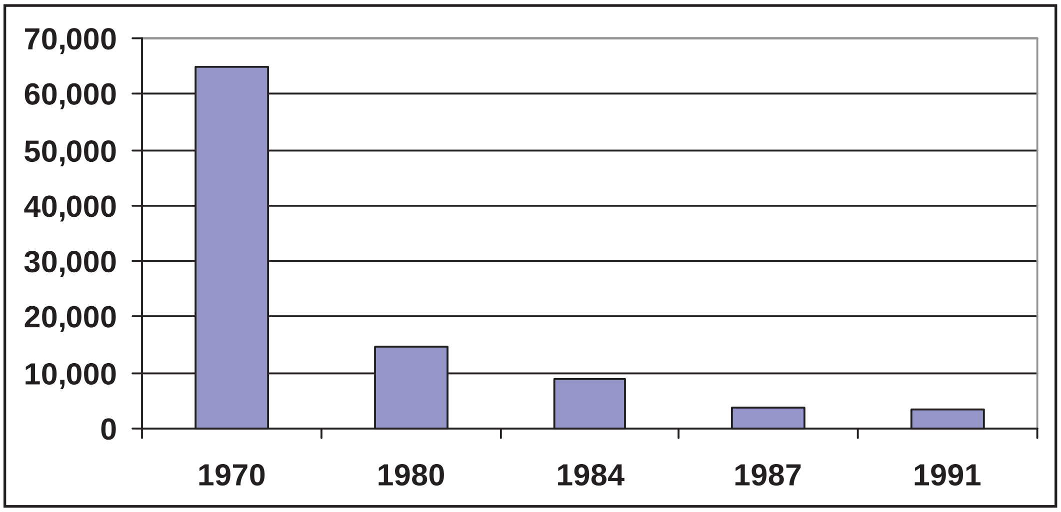


Figure 1. Black rhino population decline in sub-Saharan Africa, 1970-1991. Source Emslie and Brooks, 1999.

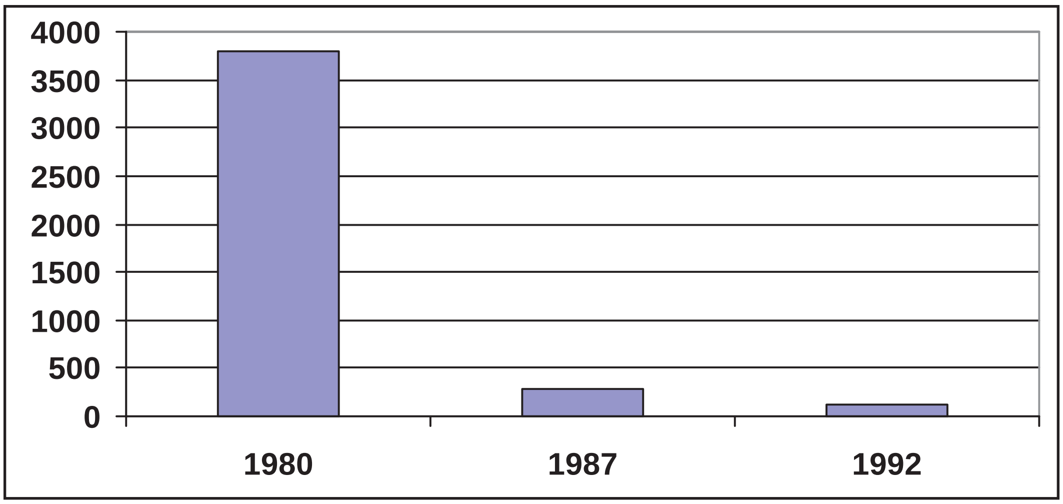
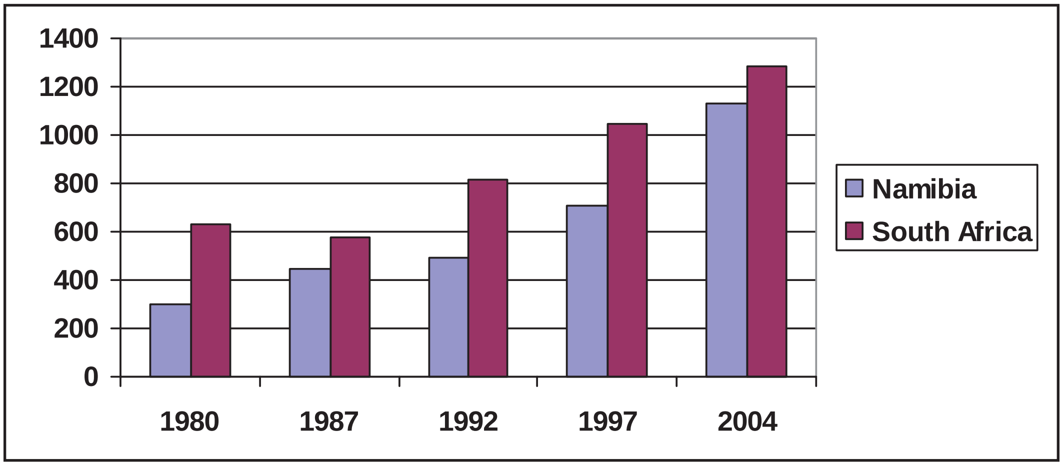


Figure 2. Black rhino population decline in Tanzania, 1980-1992. Source: Emslie and Brooks, 1999.

cial trade, as a conservation strategy (Child 2004). Namibia granted private landholders the right to manage and utilize the wildlife on their land, subject to certain regulatory restrictions, in 1967 (Jones 2001). By devolving responsibility and authority for wildlife in this way, government policies enabled landholders to capitalize on wildlife's competitive economic advantage over alternative agricultural land uses in semi-arid areas. The result was a broad expansion of wildlife populations; game numbers increased by an estimated 80% on private lands in Namibia from 1972 to 1992 (Barnes and de Jagr 1996). South Africa also developed a policy of private ownership of wildlife, and has witnessed a similar expansion of the land devoted to game species during the past thirty years. While these policy changes applied only to freehold lands, which were held primarily by white minority landowners, Namibia and Zimbabwe later spread the approach to their communal land areas as well. Namibia's community conservancies, whereby rural communities are granted the right to manage and capture the benefits from wildlife on communal lands after they have formed registered conservancies, have been particularly successful in generating local revenues and leading to wildlife population recoveries since 1998 (Jones 2001; NACSO 2004). Among other successes, the Kunene Region of northwest Namibia, where many of the community conservancies are located, is now home to the largest free-ranging black rhino population in Africa, with about 140 animals ranging across the semi-desert environment of this area's

Figure 3. Black rhino population trends in Namibia an dSouth Africa from 1980-2004. Source: Emslie and Brooks 1999, CITES (nd)b., CITES (nd)c.



communal lands (Barnard 1998; Child 2005; CITES (nd)c).

The effectiveness of these privately oriented, market-based conservation policies in southern Africa have also been demonstrated through the region's experience with the other species of African rhino, the white rhinoceros (Ceratotherium simum). This species was nearly extinct as a result of excessive sport hunting and displacement by the late nineteenth century, when the few remaining animals survived on what is now Hluhluwe-Umfulozi Game Reserve, in eastern South Africa. From that low point, white rhino numbers began a steady recovery through vigilant protection in a few reserves (Figure 4). By the late 1980's, with the population continuing to grow and the need to expand the land area available to white rhinos, government authorities began selling white rhinos to private landowners. The price of white rhinos grew steadily under this system, driven by a market for ecotourism, hunting, and additional live sales of surplus animals, and totaled $1.57 million in South Africa's KwaZulu-Natal Province in 1998 (Emslie and Brooks 1999). As white rhino numbers recovered, South Africa also reintroduced trophy hunting of the species beginning in 1968 as one economic use of the animals. From 1968 to 1996, white rhino hunts in South Africa generated a total of $24 million. In 1994, South Africa's white rhino population was transferred to Appendix II of CITES for purposes of live animal sales and trophy hunting only. Following the development of these utilization options and incentives for white rhinos on private lands, the number of rhinos held privately grew rapidly, making up 20% of South Africa's total white rhino population by 1997 (Emslie and Brooks 1999). This mixture of public stewardship and private incentives has been the key to the species' recovery to over 11,000 animals at present, nearly all of which reside in South Africa. The result of the white rhino's recovery is that this species is now the only one of the world's five rhinos that is no longer critically endangered. The key issue in the continuing recovery of the white rhino in southern Africa is the demand for rhinos by private landowners, so that the land area available to the species may continue to grow.

Black Rhino Management and CITES

By the early 1990's, as the black rhino population hit its all-time low and the species disappeared from most of its former range, conservationists and policy-makers began to reappraise their approach to international rhino trade regulation. The ninth CoP to CITES, held in 1994 in Fort Lauderdale, Florida, passed a broad resolution on rhino conservation which recognized that the trade ban was insufficient to protect and recover rhino populations (Emslie and Brooks 1999; CITES (nd)a). This resolution called for rhino range states to develop their own locally appropriate management plans for self-sufficient rhino conservation and recovery.

South Africa and Namibia had already adopted a management plan in 1989 and developed a rhino management group that worked to standardize reporting and coordinate managers.[1] Namibia later developed its own management plan for black rhinos which aims to maintain a long term viable population of 2000 animals and to implement a sustainable use scheme for generating benefits in order to support and justify the conservation of the species (Barnard 1998; Emslie and Brooks 1999).

In line with their management objectives and the continuing growth of their black rhino populations, South Africa and Namibia introduced a request to the thirteenth CITES CoP , held in October 2004 in Bangkok, Thailand, to grant them a limited number of export permits for black rhino hunting trophies. This represented the first proposal for trade in black rhinos since the Appendix I listing in 1977, and while there is an established system within CITES for granting hunting trophy export permits for other Appendix I species such as leopard, there had been no legal trophy hunting of black rhinos in Africa for decades. This made the proposal a controversial and precedent-setting one, and it faced opposition from animal welfare groups opposed to hunting on ideological grounds, as well as rhino range states such as Kenya and India which oppose legal trade in products from wildlife such as elephants and rhinos because they argue it will encourage poaching. The proposal was originally for ten export permits for trophy hunted rhinos in South Africa and five in Namibia. Prior to the conference, South Africa reduced its request from ten to five based on concerns expressed about its proposal by scientific advisory groups and conservation organizations. The conference approved the proposal, and in its resolution on the matter cited the prior COP-9 resolution instructing rhino range states to develop management plans, highlighting the potential value of sustainably managed hunting to species conservation and recovery (CITES (nd)d). In this respect, an important factor in this decision was probably the success that South Africa had demonstrated in using limited trophy hunting to help support the recovery of its white rhino population. Namibia's par-

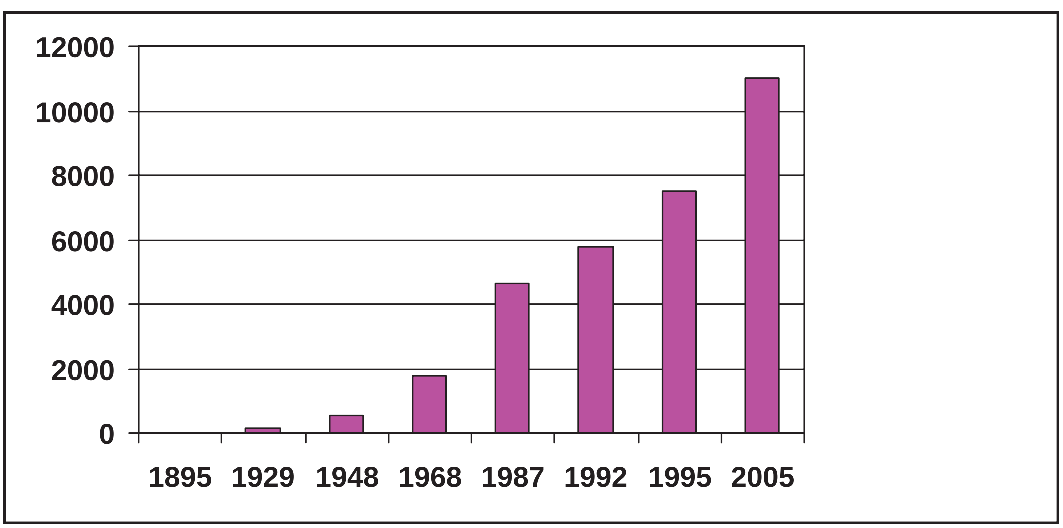


Figure 4. Southern white rhino population recovery since the late nineteenth century. Source: Emslie and Brooks 1999.

[1] Namibia was a part of South Africa until 1990, when it gained political independence.

ticularly strong record in establishing wildlife management practices benefiting local communities through its community conservancies was probably also valuable in generating support for the rhino hunting proposal. Important elements in the proposal's success were its provisions for all hunting to be of 'surplus' non-reproductive male black rhinos, and for the money generated by these hunts to be reinvested in conservation and recovery of the species.

The Namibian and South African proposals succeeded because of those countries' impressive established track record in black rhino conservation and the fact that they collectively hold the vast majority of the species' continental population. The two countries have established clear management plans for rhinos as called for by the ninth CITES CoP in 1994, and possess increasing rhino populations subject to little illegal use. Despite continued public resistance to trophy hunting as a management tool for endangered species, particularly in North America and Europe, South Africa and Namibia were able to overcome this sentiment through their successful track record and a sense among the conservation community that this success should be rewarded (Leader-Williams et al. 2005).

Trophy Hunting and Black Rhinoceros Recovery: Current Trends and Issues

The decision to allow for a limited trophy harvest of black rhinos under CITES represents a watershed in efforts to conserve and recover this critically endangered species. The move resulted from the empirically demonstrable management successes in wildlife, and specifically rhino, management on the part of Namibia and South Africa, as well as recognition of the broad failure of blanket trade prohibitions to recover black rhino populations in Africa during the past thirty years. The hope of regional conservationists and managers is that by departing from this framework of strict trade prohibition in favor of limited sustainable utilization, greater economic incentives will bolster black rhino recovery on private, communal, and state lands.

The success of the South African and Namibian proposals resulted from not only the empirical strength of their management practices, but is linked to a broader perceptible shift in international attitudes towards utilization and trade as conservation tools. During the same CITES CoP in Bangkok, a separate resolution was passed which adopted the Addis Ababa Principles and Guidelines for the Sustainable Use of Biodiversity, which were developed under the Convention on Biological Diversity. These guidelines stress the importance of sustainable use for the conservation of biological resources and call for international agreements to promote market forces and incentives which value wild species (CITES (nd)e). The adoption of these guidelines reflects the recent evolution of CITES from its more prohibitive traditions towards greater support for trade and utilization as parties look for creative and practical species recovery strategies. CoP-13 reflected this pro-use shift in some of its other key decisions, including the downgrading of Swaziland's white rhinos to Appendix II for purpose of live animal sales and the failure of Kenya's proposal to transfer Africa's lions to Appendix I.

There are several basic challenges facing the important experiment in black rhino utilization that will follow the CITES decision. While there is no question that South African and Namibian hunting outfitters will be able to sell their limited black rhino trophy hunts at relatively lucrative rates (up to $200,000 per animal has been suggested), a key for continued black rhino hunting will be demonstrating that the resultant revenues are reinvested in conservation. South Africa and Namibia will need to build on their established record of transparent and thorough monitoring and reporting in order to build support for their utilization practices in future CITES debates on rhino management. Ensuring that some of the hunting revenues directly benefits poor rural communities should be a priority.

But the foremost challenge facing this approach to rhino conservation is likely to come through future and existing impediments to legalized trade. The South African and Namibian proposal at COP-13 succeeded despite significant opposition from some conservation lobbies and animal welfare groups (e.g. Anon. 2004b). Although widely supported by conservationists in the southern African region, the CITES decision was often portrayed in western media reports as a negative one for black rhino conservation (e.g. Anon. 2004a). Utilization-based approaches to wildlife conservation still have not garnered mainstream acceptance internationally, as this negative coverage demonstrates.

A more direct obstacle for South Africa and Namibia's utilization policies is that irrespective of CITES rulings, countries may restrict or prohibit the importation of rhino trophies unilaterally. The United States has black rhinos listed on its Endangered Species Act (ESA) as 'endangered', and species of this status are normally not allowed to be imported. Namibia experienced frustration six years ago when the U.S. Fish and Wildlife Service ruled against changing the cheetah's status to allow trophy imports from Namibia into the U.S. Namibia is home to Africa's largest cheetah population and allows a limited number of them to be hunted on private lands in order to promote landholder incentive for conservation of the species (Leader-Williams and Hutton 2005). Currently black rhino trophies are not allowed into the U.S., which is the largest market of wealthy safari hunters in the world, and consequently it is possible that this will depress the returns that South Africa and Namibia can generate from their quota. It should be noted, however, that the listing of a species as endangered under the ESA, or its placement on Appendix I of CITES, does not necessarily prohibit importation of trophies of the species; both leopard and elephant trophies are regularly imported into the U.S. although permitting and importation procedures are quite extensive and contingent on approval of specific import applications.[2] There is considerable administrative latitude for allowing rhino trophy imports into the U.S. in the future without altering its ESA listed status, but there are also significant barriers to importing rhino trophies from Namibia and South Africa into the U.S. regardless of the effectiveness of rhino conservation programs in those countries Whether rhino trophy hunting programs receive this type of support, will largely be a matter of political negotiation between those in favor of sustainable management of wildlife and rhino recovery, and the influential animal welfare lobby which comprises the main opposition to trophy hunting.

Conclusion

There are several important practical implications of the CITES decision to allow black rhino trophy hunting. This decision represents the first departure from the strict prohibition on all forms of black rhino trade since the species was placed on Appendix I nearly thirty years ago. This makes the move an important experiment in rhino conservation policy, with the aim being to begin laying the basis for a market-based model for black rhino population growth and range expansion in Namibia and South Africa, as has long been

[2] For example, Jackson (2006) gives the recent case of USFWS refusal to grant approval for import of elephant trophies from Mozambique, despite the fact that elephant trophies are regularly imported from southern Africa and the species is considered to be increasing in Mozambique.

established for white rhinos in South Africa. If this experiment is successful, it will lead to further growth in regional black rhino numbers, greater benefits to landholders and state management agencies from rhino management, and thus greater investment in rhino conservation and recovery. Given the poor record of success of blanket trade prohibition as a conservation strategy during the past three decades, there is a strong imperative for such utilization-based experiments.

The decision to resume hunting of black rhinos was a highly controversial one, although primarily for ideological grounds relating to the legitimacy of hunting itself, rather than to the management or demographics of black rhinos per se. The resumption of hunting this species may reflect a growing sentiment among international conservation actors that pragmatic sustainable use, particularly in developing countries, needs to be given precedence over western ideologies. The international community, as well as influential nations such as the United States, should continue to support experimental efforts to develop market-based mechanisms for expanding black rhino populations and ranges in future policy debates and decisions.

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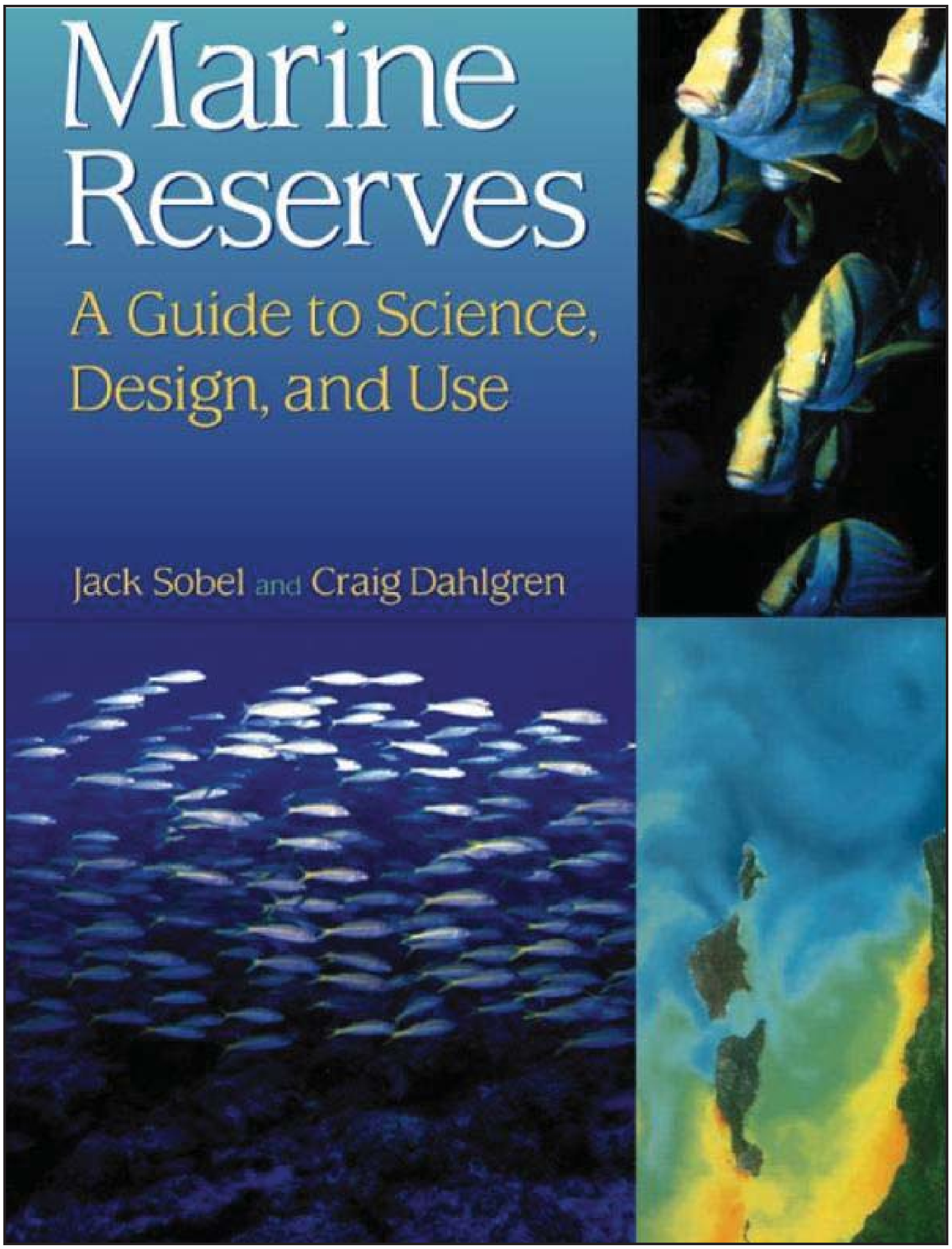
Book Review:
Marine Reserves: A Guide to Science, Design, and Use

Jack Sobel & Craig Dahlgren
Island Press 2004

Carolyn Segalini

School of Natural Resources and Environment
University of Michigan
Ann Arbor

csegalin@umich.edu



The world's oceans cover about 70% of the planet's surface, approach depths of seven miles, and are home to several hundred thousand species ranging from microscopic proportions to over 40 tons. Protecting this extraordinary system involves multiple strategies. One management tool that has been gaining interest worldwide is the concept of "marine reserves" which are "no take" areas that prohibit most extractive activities including fishing. Given its strict nature, the concept may seem quite difficult to implement, but Jack Sobel and Craig Dahlgren in Marine Reserves: A Guide to Science, Design and Use (2004, Island Press) elucidate successful examples of marine reserves around the world as well as the factors that enabled objectives to be achieved. They offer a wealth of studies with strong scientific findings on the benefits of reserves, but also stress that these reserves are not the sole answer to resolving resource issues. Appropriate design, implementation, enforcement, and the combination of the reserve concept with other management tools will help mitigate harm done to marine environments especially due to anthropogenic stresses.

While this book is written for a broad audience, it would most benefit protected marine area managers, scientific researchers, policy makers, and coastal communities interested in potentially establishing marine reserves. The goals of this book include how marine reserves can be beneficial to both the species that inhabit them and the human communities surrounding them. Other themes address an ecosystem-based approach, and the notion that marine reserves (small, large, or network-designed) will yield observable positive results.

Marine protected areas (MPAs) exist throughout the world in order to conserve natural biological and cultural resources. According to the World Conservation Union, a MPA is "reserved by law or other effective means to protect part or the entire enclosed environment." This broad definition denotes that an MPA may be multiple-use oriented and thus allow certain types of fishing or recreational activity though perhaps not others such as oil drilling or dredging. Since this term is used in different ways to define various restrictions, it is distinguished by the authors from a "marine reserve" which is an area that can potentially exist within a MPA but is wholly restrictive in that it does not allow any resource extraction including fishing. Four main benefit themes emerge in this book. These include: protecting ecosystem structure; function and integrity; improving fisheries; expanding knowledge and understanding of marine systems to enhance non-consumptive opportunities such as public awareness and education. Additional key benefits include enhancing the proportion of large-size individual species and reproductive potential. Evidence shows that the Bahamian Nassau grouper is 75% to 100% more dense in population in the Exumas Reserve relative to surrounding non-protected areas. When an area is established as a reserve, species can recover their populations with a turnaround time range of 2 to 30 years. The establishment of marine reserves also allows significant scientific research to take place because of the controlled nature of the environment, and it's protection from multiple use activities that may complicate causal explanations to events and processes. Based on scientific estimates, the establishment of marine reserves can enable an increase in biomass of at least 20% relative to unprotected areas. To this effect, the authors cite a 2002 study by Halpern and Warner in particular that reviewed 89 published studies of marine reserves that were strictly "no take" and that involved data assessment before and after reserve designation.

Chapters 1 through 3 address why enhanced protection of certain ocean areas is warranted and reviews dominant issues that have proven most harmful to marine habitats and species. Consequently, the authors identified fishing as the single most harmful activity on the ocean. The problem isn't so much in fishing itself but in over-fishing which refers to "fishing at a level that is unsustainable, causes harm or results in irreversible change." There are several subcategories of over-fishing such as "growth" or "serial", referring to individual fish that cannot grow to their maximum potential. Non-target species are also affected by fishing as they can be caught in gear as "by-catch." Trawling practices destroy the habitat that fish rely on for survival. Reef systems in particular remain highly vulnerable to destruction from over-fishing, pollution, bleaching and global climate change. Scientific studies indicate that close to two thirds of the world's reefs have been destroyed by human activities. Because reefs are home to thousands of species, their destruction has detrimental effects on other organisms. Fishing practices of all kinds and intents are addressed in this book, including commercial and recreational.

The authors note that there is an urgent need for more data assessment in various areas. The lack of information is severe enough that confidence is low about the actual conditions and degree of problems, particularly for three quarters of the fish populations in the U.S. In light of observable events like reef declines and sea urchin decimations as well as fishery collapses, a precautionary approach is advised and marine reserves should be considered in this fashion.

Chapters 5 through 7 address reserve design and research priorities. An underlying component in successful reserve design is that objectives are clear and agreed upon by all stakeholders. Ecological considerations should specifically factor in vulnerable spawning sites, larval dispersal, and general movement into and out of an area particularly by highly mobile species. In addition to ecological considerations such as habitat and species vulnerability, policy makers should include the public, especially anglers whose support and knowledge from qualitative observations are invaluable to data assessments. Monitoring is of utmost importance and should not focus solely on ecological aspects but should also assess human perceptions and values as well. Human considerations in reserve implementation should include economic effects of displaced fishers due to area closures. Opportunities should be sought to cover inevitable losses incurred.

In Part II, chapters 8 through 10 review the success and challenges of marine reserves in the United States, Bahamas, and Belize. These stories illustrate the fact that different governance regimes, cultural attitudes, and the involvement of various types of organizations all affect the design and success of reserves. What is a common theme is the importance of involving the public in reserve decisions. In the Bahamas, the Exumas Cays Land and Sea Park, one of the biggest "no take" areas in the world was established first as an MPA and then as a reserve and is managed by an non-governmental organization. Local residents were not involved in the MPA establishment resulting in a higher occurrence of poaching among the local population. Belize offers an interesting example in how diverse multiple organizations can be involved in the funding and support of reserves such as the Nature Conservancy, United Nations, and World Bank.

A concluding global review notes that reserves actually account for less than 1% of the world's oceans. Despite this, scientific evidence shows that they can be highly beneficial. The importance of having broad public participation in the process in addition to political is crucial for successful designation, management, and compliance. Strong research in combination with top-down and bottom-up policy models will enable for successful management of marine areas of concern.

This book serves as a guide for marine reserve establishment and thus does not generally focus on regulation as relates to the cyclical momentum of legislative debate nationally and internationally in establishing standards and international cooperation. Attempts in the last several years in the U.S. Congress to pass a national ocean policy encompass several resource issues including debris, pollution, overfishing and coastal development. Further discussion on the role marine reserves in national policy agendas may help to shed light on other intractable barriers to reserve establishment.

Overall, this book is very informative and enlightening and the discussion and analysis on marine reserves adds much value to the ongoing dialogue about how to most effectively protect our marine habitats and wildlife presently and in the future.

FOCUS ON NATURE ®

Just above sea level in this mesic (moderately moist) forest a small thicket of **MA'O HAU HELE**, or NATIVE YELLOW HIBISCUS, (*Hibiscus brackenridgei*) sways gently in the warm, Hawaiian breeze. Attracted to its large, brilliant yellow flowers, beneficial native bees help pollinate this perennial (active throughout the year) shrub. Growing up to 10 feet tall it usually blooms twice each year. Off in the distance the sound of munching herbivores can be heard. They, too, are attracted to this hardy hibiscus, especially when the surrounding forest starts becoming dry. Conservationists in Hawaii have helped this "official state flower" survive by building exclosures to keep out both animals and introduced plant species. However, a bramble of blackberry persists on encroaching to compete with this particular ma'o hau hele thicket. Without more intervention, we know who the winner will be. *Artwork and text by Rochelle Mason © 2003-2006 www.Rmasonfinearts.com*

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