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CONGRESS SPONSORS
In July 2012, wildlifers from around the globe gathered in Durban, South Africa for the IVth International Wildlife Management Congress (IWMC). The Wildlife Society (TWS) initiated the concept of the IWMC nearly 20 years ago. The first IWMC was held in San Jose, Costa Rica, in September 1993 (before the first TWS Annual Conference in 1994), drawing 521 participants from 66 countries—a gratifying turnout allowing an exchange of information between developed and developing nations. The second IWMC occurred in Godollo, Hungary, in summer 1999, where 357 participants gathered from 40 countries. The third took place in Christchurch, New Zealand, in December 2003, with 943 attendees from 52 nations—the largest gathering of wildlifers ever in the Southern Hemisphere. The IVth IWMC, co-hosted with the Wildlife and Environmental Society of South Africa, was also a success with nearly 400 attendees from 35 countries. The atmosphere was charged with dynamic energy from discussions of rhino poaching, tiger conservation, fragmentation by roads and canals, international models of conservation, cross border cooperation, conflict management, wildlife ranching, and contemporary concerns across North America, Europe, Africa, Asia, and the other corners of the world. Radio and newspaper journalists closely covered the event and broadcast the news widely in Japan, Germany, South Africa, and elsewhere around the world, providing powerful exposure for issues that concern us all.

As the IVth IWMC in Durban illustrated, the question of how to address human influences on wildlife is a global concern that requires international cooperation among wildlife scientists and managers. Such international collaboration has become increasingly important to The Wildlife Society (TWS). Indeed, the importance of our involvement in this arena cannot be overstated. Wildlife is, after all, an international resource, and all wildlifers should view it as such. This kind of international collaboration with partners in the host country is essential if TWS is to effectively address the mounting challenges to wildlife management and conservation. These include the human population explosion, habitat loss and fragmentation, disease emergence, the spread of invasive plants and animals, climate changes, pollution, the devaluation of wildlife through practices such as game farming, the decline or lack of dedicated conservation funding, threats to the sustainable use of wildlife, changing property rights, negative human attitudes towards wildlife, and the disconnect between humans and nature, which leads to conservation apathy. These are only some of the challenges biologists worldwide have to deal with in the day to-day management and conservation of wildlife species. International congresses bring these issues to the world stage and convince us of the importance of meaningful involvement with wildlife beyond the borders of North America.

One of the seven pillars of the North American Model recognizes wildlife as an international resource, and TWS has always acknowledged the importance of this principle and taking steps to be involved in international management and conservation. This priority is reflected by our own membership, which now extends well beyond North America to include members in 51 countries around the globe, from Andorra to Uruguay. Many of our international colleagues join us at our An-
annual Conference to share their knowledge and learn from the science, research, and fieldwork of North American wildlifers.

International wildlife management is something TWS will always be involved in. The Society has always been concerned with worldwide events related to wildlife. This remains one of our strengths, and is becoming more important than ever as human populations grow and habitats shrink. Aside from its merits for wildlife conservation, international collaboration to protect our natural resources reflects our shared humanity and enriches the human spirit—a win for all species that inhabit the Earth.

Dramatic changes in international wildlife management are as fast paced as the changes in the world of publication—the move from paper to paperless manuscripts and books. The proceedings of the first and second IWMC were published by TWS, but for the third IWMC TWS only published the abstracts due to publication expenses and rising costs of international postage. For the IVth IWMC the Congress organizers opted for on-line publishing. This is still a fairly new concept and one not embraced by all members of our profession. Thus, of the 135 oral presentations, 30 posters, and papers in workshops, panels, and symposia presented, we only received a handful of manuscripts to include in the electronic proceedings. Fortunately, they cover the globe and are representative of the papers at the Congress.

As we wrap up the IVth IWMC, plans are underway for the V IWMC in Sapporo, Japan in 2015. I encourage you to come and look forward to seeing you all there. Keep it wild!

ABOUT THE WILDLIFE SOCIETY

The Wildlife Society is committed to a world where humans and wildlife co-exist. We work to ensure that wildlife and habitats are conserved through management actions that take into careful consideration relevant scientific information. We create opportunities for this to occur by involving professional wildlife managers, disseminating wildlife science, advocating for effective wildlife policy and law, and building the active support of an informed citizenry.

Our mission is to represent and serve the professional community of scientists, managers, educators, technicians, planners, and others who work actively to study, manage, and conserve wildlife and habitats worldwide.

The members of The Wildlife Society manage, conserve, and study wildlife populations and habitats. They actively manage forests, conserve wetlands, restore endangered species, conserve wildlife on private and public lands, resolve wildlife damage and disease problems, and enhance biological diversity. TWS members are active across the United States, Canada, and Mexico, as well as internationally.

The products of The Wildlife Society include essential, practical, and objective information for wildlife professionals. We provide research, policy information, and practical tools in print and electronic forms, along with vibrant professional networks that allow solutions to wildlife conservation and management challenges to be anchored in science.
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WHAT IS THE FUTURE OF BISON CONSERVATION?

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ABSTRACT: The conservation of the plains bison (Bison bison) is considered one of the greatest conservation success stories in North America. Although the historic distribution of bison (plains and wood bison [B. b. athabascae]) was larger than any other indigenous large herbivore in North America, market hunting and competition with the livestock industry reduced the plains bison to ≤1,000, with only 25 free-ranging animals left in the world by 1902. Through the cooperation of private individuals, non-profit organizations, and the federal governments of the U.S. and Canada, bison were saved from extinction and are now scattered throughout much of their historical range, numbering >500,000 individuals. Despite the numerical recovery of the species, recent questions have surfaced regarding the true success of this effort as <21,000 plains bison (<5% of all bison) are managed within conservation herds (n = 62). Thirteen percent (n = 8) of herds are outside of their historical range, 92% (n = 57) have <1,000 individuals, and 8% (n = 5) are managed on areas of >2,000 km². Based on these data, new questions have been posed regarding the direction of bison restoration at the continental scale. Taking into account challenges associated with bison management (e.g., disease transmission, genetic introgression of domestic cattle, legal designation of bison, availability of restoration sites), we examine whether the current model of numerous small, confined bison populations represents ecological recovery of bison. We then outline recent conservation initiatives to demonstrate that a decision on the future objectives of bison conservation needs to be addressed.

KEY WORDS: bison, Bison bison, conservation challenges, conservation, management, restoration

Plains bison (Bison bison; hereafter referred to as bison) historically ranged across North American from the Rocky Mountains to the eastern seaboard and the plains of Canada to the northern reaches of Mexico (Reynolds et al. 1982, Danz 1997). In total, bison resided in 4 Canadian provinces, 42 U.S. states, and 5 Mexican states, an area of >9,000,000 km², thus encompassing the largest distribution of any indigenous large herbivore in North America (Figure 1; Gates et al. 2010b). Throughout the Great Plains, bison interacted with a host of species including pronghorn (Antilocapra americana), elk (Cervus canadensis), deer (Odocoileus spp.), prairie dogs (Cynomys spp.), wolves (Canis lupus), grizzly bears (Ursus arctos) and grassland bird species through ecosystem alterations (Coppock et al. 1983, Krueger 1986, Knopf 1996, Freese et al. 2007). Bison also play critical roles in grassland ecology through the facilitation of vegetative heterogeneity (Knapp et al. 1999, Fuhlendorf et al. 2008). Wallowing activities can lead to standing water (Knapp et al. 1999), which in turn supports numerous plant species (Collins and Uno 1983, Polley and Wallace 1986) and provides habitat for prairie amphibians (Bragg 1940, Corn and Peterson 1996). Vegetation communities are affected by bison through grazing, physical disturbance, nutrient cycling, and seed dispersal (McHugh 1958, Knapp et al. 1999). These activities influence grassland heterogeneity, supporting prairie obligate species in the tall, mixed, and shortgrass prairies (Powell 2006, Fuhlendorf et al. 2008, Gates et al. 2010a).

Prior to the arrival of Europeans in North America, the estimates of bison numbers ranged from 15 to 100 million individuals (Dary 1989, Shaw 1995),
however most estimates range from 30 to 60 million bison (Seton 1929, McHugh 1972, Lott 2002). Following European settlement, bison numbers declined rapidly as a result of market hunting by European settlers (Hornaday 1887, Isenberg 2000) in addition to competition with domestic livestock (McHugh 1972, Dary 1989, Danz 1997, Isenberg 2000). As a result, <1,000 bison were in North America by 1890 (Hornaday 1887, Seton 1929) and wild, free-ranging bison were extirpated from Canada (Freese et al. 2007) and nearly extirpated from the U.S. (Meagher 1973).

In the U.S., the loss of large bison herds led to the first major conservation movement to preserve a species on the brink of extinction (Coder 1975). These efforts were led by private individuals who established small herds throughout the Great Plains (Boyd 2003). Private herds would later form the foundation for most of the public herds (Boyd 2003). The second conservation effort was led by the American Bison Society (formed 1905) who influenced the U.S. Congress to establish public conservation herds throughout the U.S. (Coder 1975, Danz 1997). In Canada, federal conservation began in 1907 with the purchase of the privately owned Pablo-Allard herd from Montana, U.S. (Freese et al. 2007).

These conservation efforts increased the bison population, which doubled between 1888 and 1902 (Coder 1975) and increased steadily to approximately 30,000 by the 1970s (McHugh 1972). Of the 30,000 bison, half resided in conservation herds (Freese et al. 2007), defined as bison populations managed primarily for conservation rather than commercial production (Boyd 2003). In the 1980s, commercial bison production increased, resulting in >500,000 bison in North America today; yet, <5% of bison reside in conservation herds (Boyd 2003). In fact, the number of individuals in conservation herds has remained relatively stagnant since the 1930s, and today only 20,504 individuals are located in 62 conservation herds (Boyd et al. 2010). In other words, despite the dramatic increase in bison during the past 120 years, the number of free-ranging wild bison has lagged far behind.
A further examination of these conservation herds provokes questions regarding the ecological effectiveness of bison restoration at a landscape scale. Of the 62 conservation herds, 13% (n = 8) are located outside the historical range of bison and 92% (n = 57) consist of <1,000 individuals, a population size considered to be genetically viable over the long term (Gates and Ellison 2010). Furthermore, only 8% (n = 5) of herds are located on landscapes >2,000 km² (Gates and Ellison 2010) and 16% do not contain breeding age males. Ecologically, wolves are the only effective predator of adult bison, yet they are associated with only 10% of conservation herds (Gates and Ellison 2010).

As a result, conservationists are questioning whether bison are facing an ecological extinction event (Freese et al. 2007). To reverse this trend, the Wildlife Conservation Society designed a foundation for bison restoration through the Vermejo Statement (Redford and Fearn 2007) which states, “Over the next century, the ecological recovery of the North American bison will occur when multiple large herds move freely across extensive landscapes within all major habitats of their historical range, interacting in ecologically significant ways with the fullest possible set of other native species, and inspiring, sustaining and connecting human cultures.”

Expanding on this work, Sanderson et al. (2008) established a scoring matrix to quantifying the conservation value of these herds. Exceptional contributors to ecological restoration included naturally structured herds of >5,000 individuals. Herds should consist of genetically pure and disease free animals which are impacted by all natural ecological interactions including predation. Lastly, herds located on landscapes >2,000 km² are considered excellent contributors to bison recovery (Sanderson et al. 2008). Similarly, Lott (2002) hypothesized that >13,000 km² is necessary for an ecologically functional prairie landscape. More recently, Kohl (2012) examined single foraging patch sizes of bison that when multiplied by historical spatial and temporal scales equates to landscape scales similar in size to these previous estimates.

Given the current status of bison (Boyd 2003, Gates et al. 2010a) and restoration guidelines (Redford and Fearn 2007, Sanderson et al. 2008, Gates et al. 2010b), we outline the important conservation challenges facing the ecological restoration of bison today, and then discuss these challenges in light of the current model for bison conservation (i.e., numerous small, confined bison populations). In conclusion, we outline contemporary steps that are being taken to conserve bison, and provide a comment on how things need to change to prevent the ecological extinction of bison.

CONSERVATION CHALLENGES

Domestication

Domestication may permanently alter the bison genetic pool while producing significant changes to morphology, physiology, and behavior as a result of anthropogenic selection and the loss of natural selection (Freese et al. 2007). Within the commercial herds, cattle husbandry practices are common resulting in non-random selection of traits leading to docility, reduced agility, and growth performance while also altering sex ratio and age structure (Gates et al. 2010a). The elimination of mature males further influences mate competition and natural selection (Gates et al. 2010a). Domesticated bison may also pose significant issues to bison conservation if commercial animals establish cattle introgression within conservation herds when intentionally or accidentally mixed with conservation herds (Boyd et al. 2010). Furthermore, increased commercial herds may lead to the misconception that bison are no longer vulnerable to conservation issues because of demographic recovery (Gates and Gogan 2010).

Hybridization

A concerted effort to create “beefalo” through the cross-breeding of bison and domestic cattle occurred during the late 1800s and early 1900s to create a more resilient winter species while maintaining the meat production qualities of cattle (Dary 1989, Ogilvie 1979). The concept dates back to the 16th century (Dary 1989) and was actively pursued by the Canadian government as late as the early 1960s (Ogilvie 1979). These efforts have resulted in widespread domestic cattle gene introgression in the mitochondrial and nuclear DNA of bison (Halbert and Derr 2007). Today, only 8 conservation herds are free of genetic introgression (Boyd et al. 2010) and only the Yellowstone National Park (YNP) and Wind Cave National Park (WCNP) herds have had large enough samples to confidently evaluate introgression levels (Boyd et al. 2010). Of these 2, recent work has demonstrated minimal introgression in bison in WCNP (K. Kunkel, unpublished data).

Further complicating the conservation of the bison genome is questions related to historical and geographic differences. Despite low levels of cattle introgression, some public herds may contribute to bison conservation due to unique historical and geographic lineages (Halbert 2003, Halbert and Derr 2007). As a result, these lineages are important in the long-term conservation of the bison genome regardless of introgression levels and should be preserved (Boyd et al. 2010). Similarly, herds considered free of cattle introgression should be managed in isolation from hybridized herds (Boyd et al. 2010).
Disease
Nine diseases are recognized by the International Union for the Conservation of Nature (IUCN)-Bison Specialist Group as diseases of concern for bison conservation; however, only the YNP and Grand Teton National Park/National Elk Refuge (Jackson herd) herds (24% of the bison conservation population) are significantly impacted by chronic disease issues (Aune and Gates 2010).

Brucellosis.—Brucellosis (Brucella abortus) is primarily found in bovine species; however, elk play a transmission role in the Greater Yellowstone Ecosystem (Davis 1990). Primarily transmitted through oral contact with aborted fetuses, contaminated placentas, and uterine discharges (Reynolds et al. 1982, Tessaro 1989), brucellosis leads to first pregnancy abortion in > 90% of infected female bison (Davis 1990, Davis et al. 1990). Natural immunity reduces the abortion rate to 20% and ~ 0% by the second and third pregnancy, respectively (Davis 1990, Davis et al. 1990). Due to similar symptoms in domestic cattle, infected bison populations are heavily managed to minimize transmission from bison to domestic livestock (Keiter 1997), despite no confirmed cases of transmission in the wild (Bienen 2002, Cheville et al. 1998, Shaw and Meagher 2000). No highly effective vaccine is available; however, quarantine protocols and test and slaughter protocols may effectively eliminate all animals within an exposed population (Aune and Gates 2010).

In YNP, the Interagency Bison Management Program (IBMP) was adopted as a cooperative, multi-agency plan to guide bison and brucellosis management to maintain wild, free-ranging bison, while reducing the transmission risk of brucellosis to domestic cattle (USDOI and USDA 2000). Management has incorporated multiple strategies including spatial and temporal separation of bison and cattle, optimal forage utilization, use of bison as a form of livestock (Aune and Wallen 2010). Today, bison management and conservation is highly complicated because of this conservation legacy, particularly in cases of threatened species listing (see below).

Listing.—From a global perspective, the IUCN lists the bison (wood and plains bison) as “Near Threatened” (IUCN 2012). According to federal designation, bison are a “Red-Listed” species in Mexico; however, bison are currently not listed in Canada under the Species at Risk Act because of potential economic implications for the Canadian bison industry (Aune and Wallen 2010). Bison are classified as “Threatened” by the Committee on the Status of Endangered Wildlife in Canada. Similarly, bison are not listed under Endangered Species legislation in the U.S. A primary difficulty for listing in the U.S. and Canada is complication caused by the classification of hybridized animals (Boyd and Gates 2006) and the role of commercial bison producers in the numerical status of bison.

Classification.—The classification of bison as wildlife, livestock or both, is jurisdictionally dependent. Within their historical range, bison are classified as wildlife in 4 Canadian provinces, 10 U.S. states, and 1 Mexican state; however, free-ranging populations do not exist in all these areas. Where bison do exist as “wildlife”, they are typically managed within fence preserves (Aune and Wallen 2010). Outside of these areas, bison are typically classified and managed as livestock by private ownership, thus are governed by animal health and trade regulations.

Ownership.—Bison conservation within the private sector is largely a secondary objective behind commercial production. However, conservation groups such as The Nature Conservancy, American Prairie Reserve, and World Wildlife Fund have established privately owned herds focused on bison conservation. In addition, North American indigenous peoples are playing a key role in conservation because of the bison’s cultural importance and role in restoring cultural connections in addition to dietary and economic benefits. Despite these efforts, bison managed by these groups face difficult challenges such as management restrictions, insufficient funding, and litigation.

Availability of Restoration Sites
The identification of bison restoration sites has been particularly problematic because landscapes large enough to support ecologically interactive bison populations are limited. In particular, human development and habitat conversion has expanded into many areas capable of supporting large bison populations. For example, private agricultural operations can be found in almost all areas suitable for bison restoration, even on public land. Besides direct forage competition between bison and livestock, potential conflicts may include hu-
man safety issues and property damage (e.g., fencing, crop depredation). As a result, the social difficulties of translocation have increased, particularly when considering the restoration of bison to mixed-ownership and mixed-management landscapes. These areas require detailed restoration plans which provide guidelines for dealing with management issues and conflicts. In these areas, coordination among private individuals, local, state, federal, and tribal governments, wildlife agencies, conservation organizations and other concerned parties has been difficult but successful and is a necessity for long-term success.

**DISCUSSION AND REVIEW OF CURRENT INITIATIVES**

Throughout the last century it has been easier to start and maintain multiple small herds due to challenges listed above, however this trend raises questions about the efficacy of this model in the future of bison conservation. Have bison been restored ecologically? Are conservation initiatives actually meeting the restoration criteria set forth by Freese et al. (2007), Sanderson et al. (2008), and Gates et al. (2010b)? The following review of recent bison initiatives provides insight into these questions.

**Grasslands National Park**

Grasslands National Park is located on the U.S. – Canada border in southern Saskatchewan. The park was formally established in 1981 by the Canadian federal government and is currently managed by Parks Canada. Seventy-one bison were translocated from Elk Island National Park in 2005 and have since grown to approximately 350 individuals. Disease-free and genetically pure, these animals are maintained within 182 km² fenced region of the park. Current management plans will maintain the herd at approximately 800 animals; however, a range expansion may occur in coming years providing availability for a larger population. Although large predators are not found in the area, future expansion of wolves into the area may lead to overlap with the park.

**American Prairie Reserve**

American Prairie Reserve (APR) is a non-profit organization established in 2001 with the goal of acquiring and managing private land and public grazing leases to establish a fully-functioning prairie-based wildlife reserve in north-central Montana, USA. Since establishment, APR has purchased and leased 498 km² of prairie, of which 57 km² currently support bison. Approximately 325 disease-free animals currently graze the reserve. All animals have been genetically tested and all animals acquired from WCNP which contained levels of cattle introgression were removed from the population in 2011. Genetically pure animals from Elk Island National Park were used to augment the population in 2010 (n = 94) and 2012 (n = 70). The bison pasture will double in the coming years and will be further expanded as land purchases allow. No large predators currently exist in the region; however, grizzly bears have ventured within 200 km of the reserve (Robbins 2011). Future reserve plans include a landscape of >14,000 km² that support a full suite of native wildlife including 10,000 bison and large predators.

**El Uno Ecological Reserve**

The El Uno Ecological Reserve (EUER), located in Chihuahua, Mexico, is owned and managed by The Nature Conservancy. The EUER is a part of the larger Janos Biosphere Reserve which consists of a mix of private owners and ejidos (communal agriculture lands). Bison were translocated from WCNP in 2009 with 38 animals currently occupying approximately 20 km² of the EUER. The EUER has been registered as a Wildlife Management Area with the Mexican Federal Government which will permit an increase in the bison pasture and herd size. As a result, future management will maintain a sustainable herd size of 157 animals and a maximum of 210 (Laura Paulson, The Nature Conservancy, personnel communication). No sustainable populations of large predators are currently located in the region.

These examples illustrate the range of recent restoration efforts for bison. From government-based herds on public land (i.e., Grasslands National Park) to completely private conservation herds (i.e., EUER), the conservation of bison comes in multiple forms. However, when considering the criteria for the ecological restoration of bison, many programs fall short. Are these bison herds able to move freely across the landscape? Are the effort’s goals >5,000 individuals? Will these herds face historical predation? These questions seem easy to answer, but is this the model of bison conservation that will actually restore bison, ecologically, in North America?

**CONCLUSION**

As bison conservation continues forward, we as managers and conservationists understand the challenges associated with bison restoration. Additionally, we have been provided with guidelines for the successful restoration of bison (Redford and Fearn 2007, Sanderson et al. 2008, Gates et al. 2010b). After reviewing the projects above in light of ecologically restored bison populations, it leads us to speculate on the feasibility and intentions of bison restoration efforts. If we understand the challenges and issues associated with
bison restoration, shouldn’t conservation target the restoration of the species in landscapes large enough to support ecologically restored bison herds, not small bison herds maintained behind wire? If so, then we as managers and conservationists, along with the general public, and all interested and affected parties must join in the discussion on how we proceed.

If we as society desire an ecologically restored bison population, we must then initiate a shift in bison conservation which takes us away from small, isolated conservation herds and toward ecological recovery. Naturally, there will be social, political, and biological ramifications to deal with; however, these issues are not new to wildlife conservation. To proceed, we must shift our focus toward expanding existing herds that have the potential for a large ecological footprint. In regions currently lacking bison, we must focus our efforts on the translocation of bison to areas that can meet the aforementioned conservation criteria. Never again will bison roam North American by the millions, but it is ecologically, culturally, and historically important to continue the conservation of this North American icon. Thus, it is time we shift our model of bison conservation toward ecologically functional populations rather than livestock and fenced wildlife.

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WILDLIFE MANAGEMENT AND CONSERVATION IN EUROPE: TRANSBOUNDARY SOLUTIONS

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ABSTRACT: In Europe there is considerable ongoing interest in cooperation on transboundary areas concerning both conservation and management of wildlife. The available publications, discussions, and interviews with wildlife biologists indicate that transnational cooperation could be carried out on a pan-European, regional or local scale. Pan-European cooperation would tend to focus evaluation and monitoring of population numbers in birds, for example to manage conflicts between cormorants and humans. Other pan-European agreements concern the conservation of cetaceans in both the southern (Black, Mediterranean) and the north-western (Baltic, North-East Atlantic, Irish, North) seas of Europe. Regional cooperation has focused on numerical estimates of waterbird and seal populations of the Baltic Sea, the monitoring of quail populations in south-western Europe, and population dynamics and migrations of the pink-footed goose (*Anser brachyrhynchus*) between the Svalbard Islands and Northern Europe. In addition 12 countries in Eastern Europe have embarked on a joint program of vaccinating red foxes (*Vulpes vulpes*) against rabies. Local cooperation schemes include population census of large carnivores, migration of moose (*Alces alces*) in Scandinavia, a trilateral quality assessment of the Wadden Sea ecosystems, the Pyrenean network for mountain galliforms, population census and management regimes of the Dinaric brown bear (*Ursus arctos*), a recovery program for Iberian lynx (*Lynx pardinus*), and population censuses of chamois (*Rupicapra rupicapra*) in the Tatra Mountains. The level of transboundary cooperation in management and conservation is very high in Nordic countries but is unsatisfactory in many remaining countries in Europe.

KEY WORDS: birds, conservation, Europe, international cooperation, mammals, management, transboundary solutions.


The number of wildlife species whose distribution is limited to a single country is relatively low. For this reason, in Europe there has been an ever-increasing interest in cooperation on transboundary areas, concerning both conservation and management of wildlife. The management of waterbirds making seasonal migrations should be based on an international monitoring scheme for population dynamics which, in turn, would provide the foundations to determine harvest quotas for particular countries. Transboundary cooperation is also necessary for the management of ungulates in mountains, where, in winter, these animals migrate regularly into lower areas (Findo et al. 2006). In these transboundary areas, where large carnivores occur, the monitoring of the population dynamics within the prey—predator interactions is an essential element of conservation practice (Salvatori et al. 2002). In the case of marine mammals, living both in territorial and international waters, the cooperation between various countries is important in order to eliminate such factors as human disturbance, fishing methods, pollution, and maritime traffic effects which pose threats to their populations maintaining viable sizes. Thus, the objective of this study was to present a dozen or so programs which already implement the conservation and management of wildlife in transboundary areas, as well as those which could be implemented in these areas after suitable international agreements have been signed.

STUDY AREA

The land area of Europe is divided by borders of 44 countries, and the countries with sea coast have demarcated territorial waters and economic zones. The management and conservation of wildlife in transboundary areas is further complicated by differing status of wild animals which in particular countries can be national property, the property of landowners or *res nullus* (i.e., ownerless property; Bobek 1991). The cooperation in cross-border areas concerns also the waters of the Atlantic, the Mediterranean Sea, the North Sea, the Black Sea, and the Baltic Sea. The European Union of 27 countries is an organization of paramount importance to the issues of international cooperation. For all its member states, the Union has set compulsory standards concerning the conservation of wildlife and their habitats covered by the “Natura 2000” program.

METHODS

This review paper is based chiefly on materials published in both international and local scientific journals. Additional data were gathered through participation in 11 international conference symposia and congresses on wildlife-related issues which have been held in the last 2 years. During these events the data were taken from oral and poster presentations, and from formal
and informal discussions. Particularly helpful was the participation in the Pan-European Duck Symposium (Jindrichov, Czech Republic), European Congress of Conservation Biology (Prague, Czech Republic), the Hunting for Sustainability Conference (Ciudad Real, Spain), the Congress of the International Union of Game Biologists (Barcelona, Spain), and the conference on Vertebrate Pest Management (Berlin, Germany). The work was supplemented by a dozen or so interviews with wildlife biologists taking part in transboundary programs in the field of research, management and conservation of wildlife.

RESULTS

Pan-European Cooperation

*International Waterbird Census (IWC).*—This is a site-based counting scheme for monitoring waterbird numbers. Since 1967, the census has been organized annually by Wetland International (WI). The results are published (Delany and Scott 2006) and often used in the implementation of many international agreements such as the Convention of Migratory Species (CMS), and the African-Eurasian Waterbird Agreement (EAWA). In Europe, the IWC is carried out in 27 countries by ca. 2,000 people.

*Pan-European common birds monitoring scheme.*—This is a joint initiative of the European Bird Census Council and Birdlife International. Each year, birds are counted using standardized field methods (Volfíšek et al. 2008) at sample plots selected throughout a specific area of a given country. The majority of work is done by skilled volunteers and managed by coordinators (Greenwood 2007). The results are used to calculate trends in the dynamics of numbers among selected species, over long- (since 1980) and short-term time frames (since 1990). The calculations allow one to obtain supranational species indices as well as trends and multispecies indicators. In 2011, the population census was carried out in 25 countries.

*Great cormorant.*—In Europe, the data on the population number of great cormorants (*Phalacrocorax carbo*) have been very divergent, and for the years 2005—2006, they ranged from 866,000 to 1.7 million (Delany and Scott 2006). It resulted in an acute conflict with commercial fisheries and recreational angling interests. In the years 2000—2001, the European Union financed a project entitled “Reducing the conflict between cormorants and fisheries on a Pan-European scale”. The project was carried out by 49 individuals from 25 countries (Carss 2001). The final report of the project, code named REDCAFE states that there is no single solution to reduce the cormorant-fisheries conflict on a Pan-European scale. The REDCAFE report contains recommendations to all interested countries to devise a common strategy to mitigate the conflict with cormorants. Regrettably, individual states ignored the recommendations and continued their own national or regional mitigation policies, despite them not bringing the expected results.

*Agreement covering the Conservation of Small Cetaceans of the Baltic, North-East Atlantic, Irish and North Seas (ACCOBAMS).*—This agreement includes the marine environment around the shores of 17 countries. Under ACCOBAMS, 5 research projects have been implemented, concerning the effects of pollutants on the reproduction in dolphins, to develop a database concerning the situation of strandings, and the conflict between porpoises and fisheries. On the basis of the results obtained so far, and those available in publications, a new concept of conservation/management units applicable to marine mammals was formulated (Palsboll 2009). International teams conducted population censuses that included porpoises and 3 species of dolphins. They also delineated the proposed conservation/management units (Evans and Teilmann 2009). Under the auspices of ACCOBAMS, the Recovery Plan for the Baltic Harbour Porpoises *Phocoena phocoena*; (i.e., Jastarnia Plan) has been implemented (Pawliczka 2009). The ACCOBAMS is a good example of international cooperation within transboundary areas of the countries-signatories to this agreement.

*Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS).*—The purpose of ACCOBAMS is to moderate the threats to cetaceans through suitable management of environmental pressures associated with commercial and recreational fishing, shipping, tourist industry, coastal development and urban growth. The potential area covered by the extent of the ACCOBAMS includes 28 states. At present, research projects are implemented concerning the stranding sites (on shoals), population census and structure, anthropogenic noise, chemical pollution, and the impact from shipping (Evans 2008, Notarbartolo di Sciara and Birkun 2010).

Regional Cooperation

*The pink-footed goose.*—This species (*Anser brachyrhynchus*) nests in Norway, on the Svalbard Islands, but spends its winters in Denmark, the Netherlands, Belgium, and partly in Norway. The current population number is 70,000 (Trinder and Madsen 2008). During their wintering, the pink-footed goose causes considerable damage in stubble fields, winter wheat fields, and in grasslands. It is feared that a further increase in population numbers at nesting areas will cause long-term degradation of wet tundra habitats.
An international management plan was drafted for this species (Madsen and Williams 2012). The plan for the pink-footed goose presumes keeping the population level at ca. 60,000 individuals, through hunting based on the monitoring of population numbers. However, to implement this plan in practice, the consent of the relevant authorities in each state within the range of this species is required.

**The common quail.**—Between 1970 and 1990 there was a dramatic drop in numbers of the common quail (*Coturnix coturnix*). Burfield (2004) suggested that, at present, the number of common quails in south-western Europe had stabilized. To verify this, in 2005—2009 an international team conducted a monitoring scheme of population numbers in 11 breeding sites within France (3), Spain (5), Portugal (2), and Morocco (1). The analysis of the data obtained indicated that the quail population in the study area was stable (Rodriguez-Teijeiro et al. 2010). It is recommended that changes in agricultural and environmental policies should be introduced that delay mowing and lengthen the biological cycle of cereals.

**Status of Wintering Waterbird Populations in the Baltic Sea (SOWBAS).**—The objective of this program, implemented during 2007—2009 was to estimate the numbers of waterbirds wintering in the Baltic Sea area. Nine countries with Baltic Sea shorelines participated in the project. The numbers of 20 species of birds was estimated at 4.41 million (Skov et al. 2011). This estimate was 41% lower than that for a similar census undertaken in 1992 and 1993 (7.44 million; Durick et al. 1994). It is thought that the reduction in population numbers of the studied species of birds has been caused by a decline in the nutrient loads of the coastal water of the southern and central Baltic Sea, climatic changes, oil contamination from tankers, and gill nets used for fishing.

**Nordic Waterbirds and Climate (NOWAC) Network.**—Elmberg et al. (2006) stated that there is a lack of important data needed to devise management strategies for the effective sustainable exploitation of migratory waterbird populations. In 2010, a group of professional ornithologists, hunters and administrators from Denmark, Finland, Iceland, and Sweden created the NOWAC network. The aim of this group’s work will be to derive improved data on population and harvest size, recruitment and survival rates and establish a mechanism to ensure the sustainable harvest of quarry waterbirds (Fox 2012).

**Grey seal in the Baltic Sea.**—It was necessary to undertake international action for protecting the grey seal (*Halichoerus grypus*) populations in the Baltic Sea, after the seal numbers dropped from ca. 100,000 at the end of the 19th century to only 3,000 in the 1970s (Harding et al. 2007). The main reason for the declining seal population was overhunting (Harding and Harkonen 1999, Kokko et al. 1999). The restitution of the grey seal population began after the signing of the Convention on the Protection of the Marine Environment of the Baltic Sea Area in Helsinki (HELCOM), in 1974. The international census of grey seals began in 1990 and the seals are counted in all countries around the Baltic Sea in a 2-week period at the break of May and June each year. In 2006 population numbers were estimated to be about 21,000 individuals. Half of these were living in Finland, where hunting season for grey seals had been re-opened. There are international research projects concerning migration routes of seals with the use of satellite transmitters (Sjoberg et al. 1995), and monitoring habitat selection (Sjoberg and Ball 2000, Karlson 2003).

**Fox rabies control.**—In the past, rabies (Lyssa rabies) was a widespread viral disease prevalent among red foxes (*Vulpes vulpes*) in Central and Eastern Europe and helped control population numbers of this species (Anderson et al. 1981). Since rabies poses a threat to humans as well as to domestic animals, the European Commission ordered a wide-scale program of oral immunization of foxes (OIF). The objective of the program was for the each EU state to reach the status of being ‘officially’ free of rabies. This goal was achieved in all countries in Western Europe. In Central Europe the action to immunize foxes began in 2000, and is ongoing in Estonia, Latvia, Lithuania, Poland, Slovakia, Hungary, Bulgaria, Italy, Russia (Kaliningrad), Slovenia, Montenegro, and Serbia (Rabies Bulletin-Europe 2010). The vaccine is placed inside a pellet of bone-meat mix which is readily consumed by foxes. In Poland, oral immunization of foxes brought about a dramatic drop in the numbers of these animals which were carriers of rabies (Flis 2009). But the side-effect of immunization was a dramatic increase in the number of foxes and a drop in numbers of small game animals (Kamieniarz et al. 2011).

**Local Cooperation**

**Gallipyr project.**—This is a network for the conservation of mountain Pyrenean galliforms. The project is implemented in the transboundary regions of France, Spain and Andorra by the “Forespir” organization (Ayala 2011). In the course of the project, there are inventories of power line cables and wire fences being responsible for the deadly collisions with these species of bird. To reduce the number of fatalities, strips of specific colored materials were used to make these obstacles more visible to the birds. About 180 km of fences, and a third of dangerous power line cables, have already been visualized. The spatial distri-
bution of breeding habitats of capercaillies, partridges, and rock ptarmigans were also studied, with research on the impact of terrestrial predators, including wild boars, on capercaillie populations.

**Trilateral Wadden Sea cooperation.**—Since 1978, the Netherlands, Denmark and Germany have been working together on the protection and conservation of the Wadden Sea. The main tasks of the Trilateral Wadden Sea Cooperation was to identify ‘ecology targets’ and to establish a ‘trilateral monitoring program’. The ecology targets are described for 6 habitat types, and the objective of the trilateral management is to guarantee the natural functioning of the ecosystem of these habitat types through proper regulation of human activities. Trilateral Monitoring and Assessment Program was initiated in 1994, and monitoring of human activities, pollution, algae, benthos, plant community, birds, fish and seals are performed regularly (Reijnders et al. 2005, Laursen et al. 2010).

**Large carnivores in Sweden and Norway.**—The brown bear (*Ursus arctos*), Eurasian lynx (*Lynx lynx*), wolverine (*Gulo gulo*), and the gray wolf (*Canis lupus*) occur in Sweden and Norway. Past extermination campaigns reduced their numbers and led to the extinction of wolves and bears in Norway. From 2004 to 2007, the parliaments of both countries made it a political objective to restore the numbers of these 4 species, with simultaneous minimization of conflicts with humans. A series of agreements were signed between Sweden and Norway, concerning the estimation of population numbers, with the use of comparable methods (Kindberg et al. 2011, Swenson and Kindberg 2011).

**Brown bears in Croatia and Slovenia.**—Croatia and Slovenia share the same Dinaric brown bear population (*Ursus arctos*) estimated to be ca. 1500 animals (Reljic et al. 2012). The studies on brown bear populations in the transboundary area are conducted by a Croatian—Slovenian team with some participating experts from Norway. The work concerns the estimates of population numbers and the evaluation of various management regimes applied to brown bears in both countries.

**Moose displacement patterns.**—This was a joint Swedish—Norwegian research project concerning the movements of moose (*Alces alces*) within the transboundary area in the central parts of these 2 countries. The moose (*n* = 108) were immobilized from helicopters with the use of a dart gun and then equipped with GPS/GSM collars. The majority of moose captured in Sweden (87%) and Norway (67%) was classified as migratory animals displaying regular seasonal return movements to the previous places of occupancy (Bunnefeld et al. 2011).

**Iberian lynx.**—The Iberian lynx is a critically endangered cat species with a highly restricted geographic distribution, which is limited to the Iberian Peninsula (Sarmento 2009). There is a joint Spanish—Portuguese research program aimed to recover the Iberian lynx population in both countries. It is assumed that the program will progress in 2 stages. The first consists in developing a captive population of lynx, the second involves creating a free-ranging population through the reintroduction of captive animals (Barbosa and Real 2010). To achieve the first objective, the captive population will need to reach some 60—70 individuals that will constitute a breeding stock (Vargas et al. 2008). At present, in the centers where lynxes are bred, the traditional husbandry schemes are being adapted to promote the natural behaviour of these animals whilst they are in captivity (hunting, territoriality, social interactions).

**Chamois of the Tatra Mountains.**—Chamois occur in the Tatra Mountains above tree line over an area of 322 km² (Jamrozy et al. 2007). This area is divided by the 57-km-long Polish—Slovakian border and lies entirely within 2 national parks, one to each side of the border. The joint censuses taken using the total count method have been performed since 1957 under an agreement signed by the 2 parks (Zięba et al. 2004).

**DISCUSSION**

The initiatives concerning the conservation and management of wildlife in transboundary areas of Europe face a number of barriers, as they usually infringe national interests and existing legislation in particular countries. Therefore, the majority of these initiatives are limited to the studies on the estimates of numbers and migrations of animals whose ranges cross national borders. The results of these studies indicate unambiguously that ignoring the fact that some individuals have their home ranges on both sides of a border, leads to overestimates of population numbers in transboundary areas (Bishop and Swenson 2012). The principles of joint international management plans are rarely implemented in practice. Transboundary cooperation is made easier when the species in question has the status of a protected species on both sides of the border, particularly when it lives in protected areas (e.g., national parks; Jamrozy et al. 2007). It is difficult to achieve a satisfactory level of cooperation in transboundary areas when the species is classified as a game species on both sides of the border, or when the same species is a protected species in 1 country, but a game species in the other (Salvatori et al. 2002, Reljic et al. 2012).

The monitoring of population numbers of migratory birds requires the participation of large numbers of staff who, principally for financial reasons, are not
professional wildlife biologists but volunteers. For this reason the data collected by them should be verified by professional wildlife biologists, because part of this could be the result of GIGO (garbage in garbage out). Europe is in urgent need of a supra-national structure, which will coordinate management actions for waterfowl populations, as is being performed by the Flyway Councils in North America (Hawkins et al. 1984, Connelly et al. 2012). The initiative in Europe should come from the European Commission. In the member states of the European Union some issues regarding the management and conservation of wildlife are regulated by the European Commission. These primarily concern the Natura 2000 sites, with their lists of protected bird species (Birds Directive) and of protected habitats (Habitats Directive).

In summary, the level of transboundary cooperation in the field of management and conservation of wildlife is very high within Scandinavia, where it is regulated by regional international institutions such as HELCOM and the Nordic Council of Ministers. In the remaining countries of Europe the level of transboundary cooperation is low and it is only an initiative from the European Union that could contribute to improved cooperation concerning conservation and management of wildlife in transboundary areas.

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THE INSULARIZATION OF AMBOSELI NATIONAL PARK, KENYA

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ABSTRACT: Large mammals in Amboseli National Park depend on neighboring Maasai group ranches for wet season dispersal. However, the expansion of human infrastructure is reducing the size of wildlife dispersal areas and leading to the insularization of the park, threatening local conservation. To determine the spatial distribution of human settlement, we collected GPS data and mapped all homes, roads, electric fences, agricultural areas, and institutions in the 1,307 km$^2$ area surrounding Amboseli National Park. We recorded the closest distances at which wild large mammals were found from human infrastructure as an index of further wildlife displacement beyond the actual area of the infrastructure. Human infrastructure occupied 6% of the area around the park, which increased four times to 24% when we accounted for the wildlife displacement radius around human structures. We identified 20 clusters of dense human activity by eye, which covered 17% of the study area. Though the majority of the land around Amboseli appears available to wildlife, the spatial location of these clusters restricts wildlife movement and threatens to insularize the park. Insularization should be urgently addressed through negotiated initiatives with the local community to safeguard Amboseli as a critical biodiversity conservation area for Kenya.

KEY WORDS: Amboseli National Park, corridors, dispersal areas, Global Positioning System (GPS), Geographic Information System (GIS), insularization, Kenya.


Dispersal areas and movement corridors are necessary for the survival of wildlife in protected areas. To maintain viable wildlife populations in small protected areas, surrounding land must be accessible for foraging, mating, and breeding, or must provide wildlife with corridors that link suitable habitats (Newmark 1993, Burkey 1994, Wishitemi and Okello 2003). Wildlife may utilize unprotected dispersal areas when resources become scarce within protected primary habitat or seasonally available outside. Amboseli National Park is an important conservation area, supporting a high concentration of large mammals that attracts 140,000 tourists and brings in over 150 million Ksh (US$2 million) annually (Okello et al. 2001). During the dry season, wild large mammals rely on Amboseli’s swamps, which are fed by springs from Mt. Kilimanjaro and Chyulu Hills (Western 1982). However, the park is too small (392 km$^2$) to independently support all of the wildlife in the ecosystem year-round, and >70% of large mammals move into the adjacent Maasai group ranches in the wet season, when food resources become more widely available (Western 1982). Wildlife also travels through the group ranches to reach other protected areas such as Tsavo West National Park (Okello and Grasty 2008). However, the development of human structures and activities in the group ranches around Amboseli has led to the contraction of dispersal areas and the insularization of the park, threatening the future of wildlife biodiversity and conservation in the region.

The Maasai group ranches are communally-owned areas of land used primarily for livestock grazing, and the traditional land tenure system of the pastoral Maasai allowed people, livestock, and wildlife to share the same landscape (Seno and Shaw 2002). Currently, most group ranches around Amboseli are undergoing subdivision, in which individuals claim private ownership of small parcels of land to secure land from immigrants and the government and to engage in profitable endeavors such as land leasing, agriculture, and tourism (Seno and Shaw 2002, Campbell et al. 2003). The resulting human population growth and development have reduced available wildlife habitat and contributed to human-wildlife
conflict (Okello and D’Amour 2008). A 33-year study of ecologically similar group ranches in southern Kenya found a decrease of wildlife populations following the subdivision of the Kaputei ranches and an increase in populations in the communally-owned Mbirikani Group Ranch (Western et al. 2009).

Furthermore, overgrazing by Maasai livestock and intensive agriculture in semi-arid rangelands has depleted vegetation and water resources needed by wildlife (Okello and D’Amour 2008). Within the Mara-Serengeti ecosystem in Kenya, land cover change due to human activities such as agriculture has led to a >50% decline in wildlife numbers over 20 years (Homewood 2004). Meanwhile, in the Tanzanian dispersal areas of the Serengeti, land use change has been less widespread and confined to a smaller area, and wildlife numbers have remained constant (Serneels and Lambin 2001, Homewood 2004).

Human infrastructure and mosaic land uses also restrict wildlife movement within the group ranches. Species that rely on seasonal migration may experience population decline, as in South Africa, where many
protected areas have been insularized through fencing, and in the Tarangire region of Tanzania, where agriculture has restricted ungulates to wet season ranges (Thirgood et al. 2004, Bolger et al. 2008, Voeten et al. 2009). In Tanzanian parks, small, insularized parks experienced an accelerated rate of large mammal extinction compared to larger reserves (Newmark 1996). Through insularization, Amboseli is in risk of becoming an ecological island (Okello and D’Amour 2008). With shifting land tenure and use around Amboseli National Park, the size and quality of wildlife dispersal areas and movement corridors within the larger ecosystem is rapidly changing. However, there have been no baseline studies on human infrastructure in the dispersal areas, nor any recent evaluation and monitoring of encroachment around the park. A greater understanding of wildlife distribution in relation to human activities will provide insight into the status of the Maasai group ranches as dispersal areas for Amboseli, and identify present and future challenges for the conservation of the park’s wildlife.

The specific objectives of the study were to: (1) identify the location and extent of human infrastructure surrounding Amboseli National Park to determine how much space in the group ranches remains available for wildlife use; (2) determine the average minimum distance wildlife kept away from each type of human structure as an index of wildlife displacement beyond the physical area of structures; (3) identify clusters of multiple human activities and examine their spatial location and orientation in relation to wildlife to determine available corridors; and (4) elaborate a potential way forward for the viability of wildlife in the park and its dispersal areas.

STUDY AREA
The study area was part of the larger Amboseli ecosystem, which is approximately 5,000 km² and includes the national park and 6 surrounding Maasai group ranches, in the Loitokitok District of southern Kenya. This work focused on the dispersal areas covering 1,307 km² immediately around Amboseli National Park (Figure 1). This area included Olgulului—Ololorashi Group Ranch (1,231 km²) and the western part of Kimana Group Ranch (75 km²).

The study area consisted of semi-arid to arid pastoral land. Regional rainfall followed a seasonal pattern of short rains (October—December) and long rains (March—May). The study area received especially low rainfall (≤ 500 mm annually; Ntiati 2002). The region also experienced some of the hottest temperatures in the district (30°C). Although agriculture

<table>
<thead>
<tr>
<th>Infrastructure Type</th>
<th>Actual area (km², % of study area)</th>
<th>Effective wildlife displacement area (km², % of study area)</th>
<th>Magnification factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bomas</td>
<td>1 (0.1%)</td>
<td>127 (10%)</td>
<td>111</td>
</tr>
<tr>
<td>Roads</td>
<td>1 (0.1%)</td>
<td>69 (5%)</td>
<td>53</td>
</tr>
<tr>
<td>Electric Fences</td>
<td>19 (1%)</td>
<td>23 (2%)</td>
<td>1</td>
</tr>
<tr>
<td>Agriculture</td>
<td>28 (2%)</td>
<td>28 (2%)</td>
<td>1</td>
</tr>
<tr>
<td>Other Institutions</td>
<td>2 (0.1%)</td>
<td>37 (3%)</td>
<td>21</td>
</tr>
<tr>
<td>Total</td>
<td>78 (6%)</td>
<td>312 (24%)</td>
<td>4</td>
</tr>
</tbody>
</table>
had spread widely in the group ranches, the dry area immediately around Amboseli was largely unaffected by changing land uses, and pastoralism remained the primary economic livelihood in the region (Campbell et al. 2003).

The plant communities were dominated by *Acacia-Commiphora* bushland and open grassland (Githaiga et al. 2003). The amount of woodland cover was recently reduced due to bush encroachment and the expansion of agriculture (Campbell et al. 2003). Given saline sodic soil conditions and a shallow and unproductive Horizon A top soil, the shift from nomadic pastoralism to agriculture in the Loitokitok District led to severe rangeland degradation in the form of decreased plant productivity and increased erosion (McCabe 2003).

**METHODS**

We assessed human infrastructure and activities in the immediate Amboseli wildlife dispersal area in wet seasons (November 2008 and April 2009) when wildlife was dispersing from the park. A team of researchers covered the entire study area on foot. We used Global Positioning System (GPS) units (GPSmap 76CSx, 1999, Garmin, Olathe, KS) to determine the location of wild large mammals and human infrastructure in the group ranch, including bomas (i.e., Maasai homesteads), roads, and other institutions. We recorded GPS coordinates as well as shape and dimensions of each structure to calculate area.

We defined bomas as homesteads erected for living purposes by Maasai people, consisting of housing units, a central livestock pen, and a fence of *Acacia* branches. We mapped all bomas within the study area (except those within the Namelok electric fence) and assessed the status of use as occupied (current residents), unoccupied (in use seasonally but no current residents), abandoned (no longer in use), or new (recently completed or under construction, and not yet inhabited). Within each boma, we recorded the total number of housing structures and the permanence of each structure, classified by dominant construction materials: permanent (concrete foundations, metal roofs, wood or brick walls), semi-permanent (no foundation, metal roofs, mud or wood walls), temporary (no foundation, grass roofs, mud walls), or incomplete (in the process of being constructed).

When mapping roads, we recorded GPS coordinates from a vehicle at every 1 km, with increased frequency when the road curved. We classified roads based on width as main (>8 m), major (2–8 m), or minor (<2 m wide). We calculated the length of each road using ArcView Geographical Information Systems (GIS, version 3.2 for Windows, Esri, Inc., Redlands, CA).

We defined non-residential institutions as any human-made structure, excluding bomas and roads, and including schools, churches, hotels, water tanks, and other structures. We also recorded location of all agricultural plots and areas enclosed by electric fences.

The primary purpose of our wildlife large mammal sightings (including all species larger than Kirk’s dik dik, *Madoqua kirkii*) was to determine the potential displacement effect of human structures and activities, rather than to determine wildlife distribution and density. We recorded the distance from the wildlife to the location of each type of human structure (bomas, roads, institutions, or agriculture) within sighting distance (about 1 km) with a range finder. We assumed that wildlife locations from human structures were a function of human impacts rather than wildlife resource distribution or ecological determinants, and we assumed that human activities at a distance >500 m
did not affect wildlife location. Researchers walked to where wildlife was first seen and recorded GPS coordinates at the center of the group.

We spatially analyzed GPS coordinates for bomas, roads, non-residential institutions, agriculture, and wildlife using maps with GIS. We mapped the perimeter and calculated the area of each structure based on its measured dimensions and its GPS coordinates. To estimate the effective area occupied by human settlement and infrastructure, accounting for further wildlife displacement beyond the actual perimeter of structures, we added the average minimum distance that wildlife kept from each type of structure to the dimensions of each structure and recalculated area. The magnification factor for each type of structure was the ratio of the area inclusive of wildlife displacement to the actual area. We identified clusters of infrastructure based on density and distribution and identified potential wildlife dispersal routes between these clusters.

We used SPSS (version 9.0 for Windows, IBM Corporation, Armonk, NY) to conduct further statistical tests. We conducted analysis of variance (ANOVA) to determine differences in the average minimum distance that wildlife kept from boma and road types, and all types of human activity (bomas, roads, and other institutions). We used Welch’s approximate t-test to compare wildlife distance to occupied and unoccupied/abandoned bomas.

RESULTS
Human structures and activities were located throughout the group ranch (Figure 2), occupying 78 km² (6%) of the immediate dispersal area around Amboseli Park (Table 1). With estimated wildlife displacement, the area increased by a magnification of 4 times to 312 km² (24%), leaving 76% of the dispersal area available to wildlife. The largest area with wildlife displacement was by taken by bomas (10% of study area), followed by roads (5%), non-residential institutions (3%), agriculture (2%), and electric fences (2%).

There were 618 bomas around Amboseli National Park (Table 2). Most bomas (61%) were occupied. Most of the bomas in the northeastern region of the study site were unoccupied, while those in the southwest were mostly occupied. Bomas were in higher concentration along roads and watering places. The majority of structures in bomas were temporary (82%). The presence of incomplete structures within bomas (9% of all structures) indicated that boma sizes were expanding.

The road network in the study area was a total of 423 km in length (Figure 2). Roads covered an actual area of 1.32 km² (0.1%), increased over 50 times to 69 km² when accounting for wildlife displacement (5%; Table 4).

Non-residential institutions in the study area included churches, schools, electric fences, lodges/hotels, airstrips, water points, and markets (Table 3). Although agriculture covered 2% of the study area, it was limited to the slopes of Mt. Kilimanjaro and within the Namelok Fence. There was no wildlife sighted close to the rain-fed agriculture on the slopes of Mt. Kilimanjaro, implying displacement beyond viewing distance.

Although we sighted wildlife throughout the study area (Figure 3), wildlife kept away from human infrastructure and activities. Wildlife maintained a greater minimum distance from non-residential institutions (275 ± 20 m) than from bomas (214 ± 15 m) or roads (173 ± 9 m; $F_{3,64} = 10.71$, $P < 0.001$). The average minimum distance kept by wildlife from occupied bomas (Table 3) was greater than from unoccupied/abandoned bomas ($t_{93} = -2.18$, $P = 0.03$).

Human settlements and activities were concentrated in 20 clusters, which occupied a total of 221 km² (17% of the study area). Clusters along the northern part of the park were small but in close proximity, with only 3.3 km between clusters B and C, 2.8 km between C and G, and 3.6 between G and H (Figure 3). Wildlife was found between clusters, utilizing existing corridors.

DISCUSSION
Though human infrastructure occupied a relatively small area, it effectively excluded wildlife from nearly a quarter of the area around Amboseli National Park when we accounted for the distance that wildlife keeps...
from these structures. Of all infrastructure, bomas had the highest wildlife displacement magnification factor. Wildlife may stay away from bomas to avoid human-wildlife conflict and competition with livestock (Talbot 1972, Lamprey and Reid 2004). Daily activity around bomas degrades wildlife habitat, as vegetation is cleared for firewood, charcoal burning and medicine, and livestock trample soil and clear vegetation (Roberson 1996, Macharia and Ekaya 2005). Also, the vast majority of structures in bomas were temporary, and their construction requires the use of local plant materials. This habitat degradation makes areas around bomas unusable by wildlife years after the bomas are abandoned, as evidenced by the fact that wildlife still kept a distance from abandoned bomas.

The Maasai are traditionally nomadic, moving with the rains to find optimal pasture, which is why many of the bomas were unoccupied (Talbot 1972). However, such movements are becoming difficult with sedentarization, increasing human population, and privatization of land throughout Kenya (Talbot 1972, Macharia and Ekaya 2005). As demand for space and resources rises, over-utilization leads to environmental degradation and decreased range productivity, resilience, and stability, which results in livestock and wildlife mortality, as observed during the study (Ogolla and Mugabe 1996).

We saw no wildlife near the Kilimanjaro rainfed-agriculture cluster, where human—wildlife conflict in the form of retaliatory poisoning, spearing, or snaring displaces wildlife (Mwale 2000, Okello and Kiringe 2004). Irrigation-based agriculture was limited to a small area within the Namelok electric fence. Although the fence was intended to eliminate human-wildlife conflict, fences displace conflicts, exclude wildlife from water resources, and provide physical barriers to movement (Worden et al. 2003, Newmark 2008, Okello and D’Amour 2008). Given limited rainfall and poor soils, the rest of the study area is unsuitable for agriculture, so it is unlikely that agriculture will expand as it has in many of the group ranches in the Amboseli ecosystem (Campbell et al. 2003).

Although roads occupied <2 km² in area, the road network was extensive and acts as a barrier to movement for some wildlife species. Most roads were poorly maintained and prone to erosion, which has led to the creation of parallel minor roads that further degrade the rangelands. Although wildlife was most...
often found near roads, this pattern is related to the fact that the areas around roads were more extensively sampled due to easy access and is not representative of the actual distribution of wildlife. Wildlife avoids roads due to noisy traffic, potential harassment by tourist vehicles, and the risk of fatal accidents. New roads should be established only when absolutely necessary and should be designed to minimize habitat fragmentation and maintained to prevent off-road driving. It is also important to consider the placement of roads in dispersal areas because they encourage settlement by increasing accessibility; bomas were most often found near roads.

Though most of the study area remains open to wildlife, the spatial orientation of human development may block dispersal from Amboseli into the group ranches, thus insularizing the protected area. Human settlement has increased along the perimeter of the park due to the relative availability of water and plant resources in the park, and an increase in the market for ecotourism has led to the expansion of tourism infrastructure and year-round settlements. The spatial distribution of clusters within the dispersal areas may also block potential wildlife movement corridors between Amboseli and Tsavo West, as similarly observed in the nearby Mbirikani and Kuku group ranches (Okello and Grasty 2008).

This study identified existing movement and dispersal routes that wildlife is currently using, which are critical to maintain if the Amboseli ecosystem is to remain viable (Sawyer et al. 2009). The clusters of human activity along the northern, eastern, and southern borders of the park have narrow corridors between them, and the presence of new bomas indicates the expansion of settlement clusters. Given human immigration in the area and the growing Maasai population, it is likely that these clusters will grow together, closing any remaining gaps for wildlife dispersal in the absence of land use planning. Increased wildlife residence time in Amboseli will put more stress on the small park, leading to habitat degradation and compromising Amboseli’s role in biodiversity conservation. Furthermore, human-wildlife conflicts will likely increase at the edge of the park.

**MANAGEMENT IMPLICATIONS**

African rangeland ecosystems cannot survive within isolated protected areas and instead rely on conservation-compatible local land uses across a large area (Wishitemi and Okello 2003, Homewood 2004). A land use plan, developed by local communities and other stakeholders, should confine the development of permanent settlements, infrastructure, and tourism infrastructure to designated existing clusters. Agreement between landowners and Kenya Wildlife Service (KWS) could set aside areas for use by pastoralists and wildlife (Seno and Shaw 2002, Ferraro and Kiss 2000). Group ranch members must determine acceptable stocking rates and carrying capacity to restore range quality and reduce animal and human mortality in the ecosystem during stressful times (Norton-Griffiths 1996). Land and natural resource use plan should be managed locally to maximize transparency and accountability (Wishitemi and Okello 2003). Locals now view wildlife as a state resource, but empowering them to participate in conservation will help increase tolerance for wildlife ranging.

Creating opportunities for local communities to reap tangible benefits from wildlife will integrate conservation and development efforts (Hackel 1999). Revenue sharing between KWS and group ranches has been mismanaged, but a revision of this program may promote an improved relationship between the park and the group ranches, as seen in Mbirikani Group Ranch, where revenue sharing has generated a positive attitude towards wildlife among ranch members (Groom and Harris 2008). Subdivision may provide new opportunities for the Maasai to benefit from wildlife, legitimizing wildlife conservation as a land use option that will provide income to rural communities. The shift from the group ranch system to individual ownership may encourage local communities to invest in ecotourism enterprises or set aside tracts of land for wildlife, which would provide income and employment opportunities (Okello 2005a, Ogutu 2002). Direct payments through established endowments can be also explored within the Amboseli ecosystem (Ferraro and Kiss 2000, Bulte et al. 2008). For example, the Wildlife Foundation in Kenya has secured migration corridors on private land through conservation leases at US$4 per acre per year in Kitengela around Nairobi National Park has achieved some success, amidst challenges of local faithfulness to contracts and fundraising issues (Ferrarro and Kiss 2000). However, any compensation arrangements should aim to improve the equitable distribution of tourism benefits and access to critical resources.

Well-designed and community-supported wildlife conservation areas have the potential to spread tourism revenue locally (Okello et al. 2003, Okello 2005a, Carter et al. 2008). Tourists may be encouraged to visit private protected areas that allow unique tourism events (e.g., hiking, horse travel, night game drives, cultural interactions with the Maasai) that are not traditionally allowed in protected areas like Amboseli National Park (Okello 2005b). For example, the Kimana Community Wildlife Sanctuary brings > 7 million Ksh to the community each year (Okello 2005b). Wildlife
dispersal from the park would be critical in supporting these conservation areas, and the community would therefore have an incentive to keep dispersal areas and movement routes open. The ongoing process of establishing community owned conservation areas within the Amboseli ecosystem should borrow heavily from studies like this so as to ensure that the sites chosen complement conservation objectives.

In December 2008 and May 2009, we presented the results and implications of this study to group ranch officials and other community members and advised them to incorporate the above recommendations into future group ranch plans (Caro et al. 2009). Olglulului-Ololorashi Group Ranch will soon begin subdivision, which will take >15 years, enough time to incorporate conservation goals into planning. To ensure appropriate action to limit the effects of human development on wildlife dispersal areas, future research should examine the use of the identified dispersal areas and movement routes by wildlife, the effects of rangeland degradation on wildlife, habitat availability and suitability, and trends in human population and development and human settlement cluster expansion. The Center for Wildlife Management Studies, where this study was conducted, will seek to incorporate such questions into its long-term research plan.

ACKNOWLEDGEMENTS
We would like to thank the School for Field Studies and the Center for Wildlife Management Studies for providing the equipment and resources for this study. We are grateful for the cooperation of the Olglulului—Ololorashi and Kimana Group Ranch officials. John Kioko assisted with with spatial analysis. We would like to acknowledge research assistance from Daniel Kaaka, local guides, Kenya Wildlife Service, and the SFS—CWMS student research teams for Autumn 2008 and Spring 2009.

LITERATURE CITED


HABITAT ASSOCIATIONS OF PERSIAN WILD ASS (EQUUS HEMIONUS ONAGER) IN QATROUYEH NATIONAL PARK, IRAN

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ABSTRACT: Persian onager (Equus hemionus onager) is an endangered species whose populations in semi-arid ecosystems of Iran have continued to decline during the last few decades. The basic ecology of onagers in Iran is poorly understood and there is scarcity of knowledge to base management activities for conservation of the species. Between autumn 2009 and summer 2010 the location of onager herds were recorded using GPS in Qatrouyeh National Park (QNP). Habitat variables, including NDVI, slope, elevation, aspect, distance to nearest road, and distance to nearest water resource, were extracted for GPS locations of onagers. Principal Component Analysis was used to determine the most important factors influencing habitat use by onagers. In the cold season (Dec—Mar), use of habitat was positively related to distance to nearest road and distance to nearest water resource, and negatively associated with NDVI. In the hot season (Aug—Nov), the pattern of habitat use was associated positively with NDVI and slope, and negatively with distance to nearest water resource. Onager habitat use was related positively to slope and distance to nearest road, and negatively to distance to nearest water resource during the moderate season (Apr—Jul). Our results showed onagers were not near water resources during cold season when water stress was least, whereas they were near water resources during moderate and hot seasons because of increased water stress. Also onagers were far from roads likely to avoid from human presence. Onagers tended to use higher slopes during moderate and hot seasons to avoid high temperatures in the plains. During the hot season when herbaceous forage was scarce in QNP, onagers used areas with high NDVI and higher cover of green bush and shrub layer. They use areas with low NDVI during cold season when annual grasses were more common after rainfall in these areas.

KEY WORDS: GIS, habitat modeling, NDVI, Persian onager, seasonal habitat use.


Horses, zebras and asses, members of the family Equidae, once flourished and inhabited a range of environments in the Americas, Europe, Africa, and Asia. Today, 5 out of 7 extant Equids are now listed as “Threatened” by the International Union for Conservation of Nature (IUCN), including Equus ferus and E. africanus as critically endangered, E. hemionus and E. grevyi as endangered and E. zebra as vulnerable (IUCN 2008). Once distributed from China to Turkey and India to Kazakhstan, the Asiatic wild ass or onager (Equus hemionus) now exists only in parts of China, India, Mongolia, Kazakhstan, Turkmenistan, and Iran (Moehlman 2002). The Persian onager (E. h. onager), 1 of the 5 subspecies of Equus hemionus, has been declining in numbers over recent decades and is restricted to 2 isolated populations in 2 semi-arid ecosystems of Iran. Consequently, this taxon was categorized as critically endangered (Tatin et al. 2003, IUCN 2008). Also, other subspecies are listed as threatened by the IUCN except for Equus hemionus hemippus which is already extinct.

One of the central tenets of behavioral ecology is that features of the environment shape animal behavior (Krebs and Davies 1997). Choice of habitat affects resource use, which in turn affects competition, grouping behavior, mating activities and ultimately reproductive success and fitness (Lawes and Nanni 1993). Understanding the forces that shape habitat use (Henley et al. 2007, Nowzari et al. 2007) is essential for understanding how behavior influences population and ecological dynamics as well as resource availability (Sibly and Smith 1985). By deciphering the rules determining
Figure 1. (A) Digital elevation model map of Bahramegoor Protected Area and its location in Iran; Black lines show the boundaries of Qatrouyeh National Park (QNP). (B) Slope map of QNP. (C) Aspect map of QNP. (D) Roads map of QNP. (E) Water resources map of QNP.
how environmental features shape behavior, conservation biologists should be able to manipulate this link by changing human behavior to improve a species’ prospects of survival, enhance ecosystem function, and improve human livelihoods in environmentally sustainable ways (Rubenstein 2010). Overall, behavioral ecology has much to offer for solving conservation problems (Caro 1998).

Currently, Qatrouyeh National Park (QNP) and Touran National Park (TNP) are the last strongholds for Persian onager in Iran (Ziaee 2008). However, except for a few studies (Groves 1974, Harrington 1977) information about the ecology of populations in these protected areas is limited. Understanding temporal and spatial patterns of habitat use provides a critical foundation for management (Moore 2008), especially in equids (Moehlman 2002). There is a lack of information about population size, habitat relationships and population structure of the Persian onager (Duncan 1992, Moehlman 2002). The aim of this study was to model the habitat use of Persian onagers to be a base for further studies which can assist in the species conservation and management (Suring and Vohs 1979, Hemami et al. 2004, Sundaresan et al. 2007, Kaczensky et al. 2008).

STUDY AREA
Bahramegoor Protected area (BPA) was established in 1972 and is 408,000 ha in size. It is located 55 km away from Neyriz city and is near Qatrouyeh town in Fars province (Darvishsefat 2006). Qatrouyeh National Park was deemed by the Department of Environment (DOE) as the core zone of BPA in 2007, which is located at 29° 10’ to 29° 26’ N and 54° 36’ to 54° 48’ E (Fig. 1). Qatrouyeh National Park has an area of 32,576 ha, and a range in elevation of 1,680—2,787 m above sea level. Its terrain consists of mountains and desert-like plains. Its average annual rainfall ranged from 150 to 250 mm per year and thus is considered a semi-arid ecosystem. Maximum temperature was 44°C and minimum temperature was –1°C during the hottest and the coldest months. Qatrouyeh National Park includes part of Zagros Mountains and consists of 3 main landscapes: the Koohsorkh-e-bozorg Mountains, the Rigjamshid, Einaljalal and Dehvazir plains, and the remaining area containing many hills and valleys. Water comes from natural springs and human-made wells and troughs. Three vegetation communities predominated: Artemisia—Zygophylum, Artemisia—Amygdalus, and Rock (Qatrouyeh NP—Bahramegoor PA comprehensive management plan 2010). Apart from Persian onager, large herbivores such as wild sheep (Ovis orientalis), wild goat (Capra aegagrus), and Jebeer gazelle (Gazella bennettii), and carnivores such as grey wolf (Canis lupus), golden jackal (Canis aureus), red fox (Vulpes vulpes), striped hyaena (Hyaena hyaena), caracal (Felis caracal), and leopard (Panthera pardus), occur in the area (Darvishsefat 2006). The number of onagers has been estimated to be 270 individuals (Hemami and Momeni in press).

METHODS
Field Data Collection
We recorded the latitude and longitude of all of Persian onager sightings in QNP during 4 seasons (autumn 2009, winter, spring and summer 2010). We drove fixed loops during day from sunrise to sunset 3 days per week—one day for each plain (Rigjamshid, Einaljalal, and Dehvazir plains)—during nonconsecutive 2-week periods every month. This resulted in 6 observations every month, 18 per season and 60 per year, excluding February and June. The research team consisted of 4 observers, each a wildlife expert equipped with 8×30 binoculars, who traveled along all available routes.

<table>
<thead>
<tr>
<th>Season</th>
<th>NDVI</th>
<th>Slope</th>
<th>Distance to nearest roads</th>
<th>Distance to nearest water resources</th>
<th>Cumulative percent</th>
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<tr>
<td>Cold</td>
<td>−0.60a</td>
<td>0.81</td>
<td>0.86</td>
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Table 1. Habitat use model of Persian onager, based on a principal components analysis.

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aEigen values
through QNP, stopping wherever a herd was spotted. To count the onagers in each herd, 1 team member approached the herd, guided by 2-way radio from the other team members who observed from a distance. When within 10 m of the herd, the GPS location was recorded along with herd size, habitat type, time and date, recently left traces of the herd like feces, urine, and spoor tracks, and water resources near the herd.

NDVI
Normalized Difference Vegetation Index (NDVI) values were used to characterize vegetation. Typically NDVI indicates the greenness of vegetation, or vegetation quality. We examined the NDVI values of the GPS locations of onagers to determine the features of the vegetation in areas used by onagers across seasons. To determine selectivity of habitat use the values associated with observed onager locations were compared to values associated with 200 random points using ArcGIS 9.3 (Esri, Redlands, CA) and Hawth’s Tools (http://www.spatialecology.com/htools, accessed 31 October 2012).

Habitat Use Analysis and Model
Six habitat variables, including NDVI, slope, elevation, aspect, distance to nearest road, and distance to nearest water resource, were extracted for GPS locations of onagers (Figure 1) with ArcGIS and Hawth’s Tools. Principal Component Analysis (PCA) was used to find the most important factors affecting habitat use by onagers. Months were classified into 3 seasons based on environmental harshness: hot season (Aug—Nov), moderate season (Apr—Jul), and cold season (Dec—Mar). Initially, we used elevation and aspect, but those variables overwhelmed the relationships with the remaining variables, which we believed were more important to describe the habitat from the onager’s perspective.

RESULTS
A total of 682 herd locations were recorded throughout the 1-year period, and the results from the PCA of those locations are in Table 1. Our findings show that in the cold season, use of habitat was positively related to distances to nearest road and to nearest water resource, and negatively associated with NDVI. In other words, onagers used areas that were farther from water resources and roads and had low NDVI. In the moderate season, onager habitat use was positively related to slope and to distance to nearest road, and negatively associated with distance to nearest water resource. That is, onagers used areas at higher slopes, farther from roads, and nearer to water resources. In the hot season, habitat use was positively associated with NDVI and slope and negatively associated with distance to nearest water source. In other words, onagers used higher slopes, and areas that had high NDVI and were nearer to water resources.

DISCUSSION
This was the first study in Iran to characterize patterns of seasonal habitat use of the Persian onager over consecutive years. Spatial and temporal variability in vegetation and habitat use is common for many free-ranging herbivores across the world (Palmer et al. 2003) and our findings show that the Persian onager is no exception.

Our results indicate that during cold season onagers were not near water resources. In cold weather the loss of water through evaporation decreases and hence the need for visiting watering points decreases too. Also, onagers used areas with lower NDVI, likely because they use areas with newly grown annual grasses in this season which, compared to the remaining areas with denser bush and shrub cover have lower NDVI values. Moreover, onagers were far from roads likely to avoid from human presence.

Our results also show that during moderate season onagers tended to use higher slopes, likely to avoid the high temperature in the plains than hills, valleys, and mountains. As with the cold season, onagers were far from roads likely to avoid from human presence. In contrast, onagers were near water resources, likely because they needed more water during the warmer and more arid moderate season.

Our results also reveal that during hot season onagers were seen in higher slopes. Lack of rainfall and reduced abundance of annual grasses in QNP likely caused onagers to use areas with higher cover of green bush and shrub layer with a higher NDVI signature. Owing to these reasons as well as needing much more water, they were also found near water resources in this water-limited season.

The results of our habitat use model accord with the results of studies performed on Persian onager’s population in TNP: onagers tend to live around hills, close to springs and water resources, and avoid roads (Madani 2008, Bagheri 2011). In addition, our findings support the results of (Momeni 2009), that there is a negative relationship between Persian onagers’ pellet groups in QNP and distance to water resources, which in turn indicates onagers’ high dependence on water particularly in the hot season.

MANAGEMENT IMPLICATIONS
The results of this study can contribute to onager management and conservation in arid ecosystems of Iran. Reintroduction programs of onagers from QNP to other-
er protected areas are under study by the DOE. Our findings indicate that habitat use is mostly affected by the availability of quality food and water but that the impacts of these key resources vary by physical factor of environment and human presence. In general, hill-valley habitats are used by onagers in QNP. Therefore, future translocations of onagers should include these habitats. The presence of onagers near water and in areas with high-quality vegetation within QNP shows that both adequate water resources and high-quality food must be abundant in any future locations. Thus, choosing future sites that maximize onager survival must take precedence if new populations of onagers are to be sustained.

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STEWARDSHIP: THE ROLE OF RURAL RESIDENTIAL ESTATES IN NATURE CONSERVATION IN SOUTH AFRICA

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ABSTRACT: The future of conservation in South Africa lies in the hands of private landowners, who need to recognize areas of high conservation value and to strive towards maintaining and enhancing the biodiversity, ecological processes and historical and cultural heritage of their lands. Kudu Private Nature Reserve (KPNR) is a 4,872 ha reserve located in Mpumalanga Province, South Africa. The KPNR is owned by 125 shareholders whose mission is to conserve the unique ecosystem for future generations. The KPNR is currently in the final phase of proclamation as a Private Nature Reserve and is supported by the Manager Protected Areas Expansion of Mpumalanga Tourism and Parks Agency (MTPA) because of the large unfragmented vegetation communities, remnants of Lydenburg Thornveld, several endemic, rare or threatened species, breeding projects with rare species, special features like the Lydenburg waterfalls, and several sites of cultural and historical importance. A detailed management plan has been completed including animal population monitoring, biannual veld condition evaluation, burning programs, alien plant control and research on rare and endangered species. The biodiversity stewardship program of KPNR aims to promote the exchange of ideas with other privately owned nature reserves.

KEYWORDS: biodiversity, conservation, Mpumalanga, nature reserve, South Africa, stewardship


Kudu Game Ranch (KGR) has existed as a share block game farm since the mid-1990s and is currently managed for conservation. Previously cattle ranching, commercial hunting and cropping had been undertaken on KGR. The attraction of KGR as a low density rural residential eco-estate is its proximity to Johannesburg and Pretoria, as well as its malaria-free status. The KGR is owned by 125 shareholders with a common aim of conserving its unique ecosystem for future generations. The KGR encompasses 4,872 ha and is located approximately 15 km northwest of Lydenburg in Mpumalanga, South Africa. (Figure 1) The Dorps and the Spekboom rivers and numerous tributaries traverse the KGR.

Several shareholders as well as visiting conservation authorities and ecologists have recognized the uniqueness of KGR, with the farm being positioned at the transition between Highveld Grassland and Mountain Bushveld biomes. Elevation ranges from 1,000 m in the Spekboom Valley in the northeast to 1,640 m on the Highveld grasslands in the west.
This elevation gradient has resulted in a diversity of vegetation types, from grasslands in the east and south, to mountainous Bushveld in the north, Thornveld in the central area and protea (Protea spp.) covered grassland slopes in the west. In the mountainous northwestern portion, several springs result in some deep valleys with perennial streams and riverine forest. A man-made, canal-fed wetland system on the south contributes to the biodiversity. The wide diversity in vegetation types contributes to the overall biodiversity on KGR including more than 40 mammal species, 335 bird species, 60 tree species, innumerable grass types, reptiles and amphibians. There are several species on the farm which are on the International Union for Conservation of Nature (IUCN) Red Data List. Several cultural and archaeological sites previously occupied by Iron Age and Boer settlers are dispersed across KGR. Central to KGR is the Lydenburg Waterfall with spectacular views.

The above special characteristics, as well as the ever present threat of mining, led a few shareholders of KGR to investigate nature reserve proclamation under National Environmental Management: Protected Areas Act 57 of 2003. Because KGR had been managed for conservation it was agreed that the added support and input from a nature conservation authority (i.e., Mpumalanga Tourism and Parks Agency [MTPA]), with a management plan containing specific objectives would give the management of KGR criteria with which to monitor change over time and to implement corrective strategies if required. In the process of proclamation, KGR would change its name to Kudu Private Nature Reserve (KPNR). The process of obtaining Nature Reserve status is described below.

**METHODS**

**Site Assessment**

After shareholder consensus, the Manager of Protected Areas Expansion of the MTPA was approached about the feasibility of incorporating Kudu Game Ranch into the Mpumalanga Biodiversity Stewardship program and the possibility of protected area status under the National Environmental Management Protected Areas Act 57 of 2003. The hierarchy of this program (Figure 2) and the categories are as follows:

- Nature Reserve
- Protected Environment
- Biodiversity Management Agreement
- Conservation Area

The landowner’s obligations as well as the nature authorities’ support increases up the hierarchy of the stewardship categories. Ecologists with the MTPA visited the KGR to perform a site assessment to determine the biodiversity value of the proposed stewardship area, whether the proposed stewardship area warranted incorporation into the Biodiversity Stewardship Program and to establish the preferred stewardship category.

**Establishment of a Stewardship Category**

Findings of the site assessment were reviewed by the Protected Areas Expansion Committee (PAEC), a panel of representatives from relevant sections within the MTPA, who decide on the suggested stewardship category of a proposed site. The PAEC determined that KGR was eligible for Private Nature Reserve status based on the many attributes worthy of conservation, namely large unfragmented vegetation communities, several endemic, rare or threatened species, breeding projects with rare species (disease free buffalo [Syncerus caffer], and sable antelope [Hippotragus niger], special features like the Lydenburg waterfalls, and several sites of cultural and historical importance. A few IUCN Red data species including Red-billed Oxpecker (Buphagus erythrorhynchus), Black Bellied Bustard (Lissotis...
The farm is home to various interesting species, including melanogaster, Southern Bald Ibis (Geronticus calvus), Long White Squill (Urginea epigea), Blue Squill (Scilla natalensis; an important plant within the traditional medicine trade), Bushman’s tea tree (Catha edulis) and the endemic, Transvaal cabbage tree (Cassonia transvaalensis) are also found on the farm. Based on habitat modelling by the Mpumalanga Biodiversity Conservation Plan, the Golden Mole (Amblysomus hottentotus meesteri) may be present on the property.

The MTPA emphasized that KGR is a largely untransformed and unfragmented natural area embedded within a larger untransformed area and contributes to conservation targets for Lydenburg Thornveld and Ohrigstad Mountain Bushveld, both of which are poorly protected in the region. The MTPA determined that 21% of KGR was important for conservation and would complement the Mpumalanga Biodiversity Conservation Plan (MBCP; Figure 3). Highveld grasslands, protea covered mountain slopes and steep valleys with forest add to the biodiversity. Because of the large size of the property, the ecological processes are relatively intact and add additional significance to the site. The diverse arrays of vegetation types located within one area managed with conservation objectives as the basis are indeed special conservation-worthy attributes of KGR. Furthermore, because KGR is located in a transition zone between grassland and woodlands, conservation of the area will provide long-term monitoring sites to gauge the effects of climate change on grass:woody plant ratios. The Lydenburg Falls are central to the spectacle of KGR and several tufa waterfalls also occur on the reserve. Finally, Iron Age (ca.1500 AD) as well as more recent Boer settlements (ca.1850 AD) and artefacts are scattered across KGR and contributed to the preservation of the cultural and historical landscape.

Agreement with MTPA

At an Annual General Meeting the shareholders of KGR passed a special resolution to apply for the status of Private Nature Reserve under the Biodiversity Stewardship program of the National Environmental Management: Protected Areas Act 57 of 2003 understanding fully that a high level of landowner commitment to conservation would be involved. The KGR Board signed the agreement between KGR and the MTPA. The basic tenet of the agreement was that KGR would be managed with biodiversity conservation as a priority and the MTPA would offer assistance where possible.

Drafting of a Management Plan

A consultant ecologist was appointed to develop a detailed management plan for KGR under the ambit of the National Environment Management Protected Areas (NEM: PA, Act 57 of 2003). The Act contains the following set of requirements for Protected Area Management Plans (including Nature Reserves). Section 41 (1) states that, “The object of a management plan is to ensure the protection, conservation and management of the protected area concerned in a manner which is consistent with the objectives of this Act and for the purpose it was declared.”

The management plan is required to determine the land use pressures and threats and to establish a baseline for evaluation of management effectiveness and comprises terms and conditions of any applicable biodiversity management plan. The following management programs consisting of tasks, responsibilities, deliverable time frames, and measurement and performance criteria are listed in the management plan:

- Animal population monitoring by aerial game counts to be performed biannually, in order to establish total numbers and where possible, sex ratios for each species.
- Biannual veld condition evaluation, taking into account the grass and tree status and composition. In conjunction with the game census, this will help to determine the optimum numbers of bulk grazers, selective grazers, browsers and mixed feeders that KGR can accommodate to maintain or even enhance the biodiversity.
- Burning programs to control bush encroachment and remove moribund and unpalatable grasses.
- Alien plant control in association and cooperation with public organizations such as Working for Water.
- Monitoring and control of water pollution.
- Erosion control to rehabilitate eroded areas. Before KGR became a game farm it had been utilized as a cattle ranch and some of the effects of overgrazing are still visible.
- Development of management capacity.
- Identification of research opportunities in liaison with other conservation bodies and academic institutions.

Zoning descriptions are included in the management plan, indicating the conservation objectives and which activities may take place in different use zones of KGR (Figure 4). The management plan also includes a conservation budget specifying allocation of funds to the various tasks and projects.

An adaptive management approach is prescribed in the management plan, whereby set targets are evaluated annually or biannually, management activities are maintained if predicted outcome is achieved, or modified and re-analyzed if not achieved. Within each management program the indicators will assist in measuring effectiveness, success or otherwise of the management actions taken. This forms the basis of an annual biological audit. The management plan was accepted by the MTPA in July 2011.

**Lodging of Zonation Diagrams with the Surveyor General**

In consultation with a surveyor and a lawyer, the various title deeds were annotated and submitted to the Surveyor General for inclusion in the Nature Reserve proclamation diagrams. The Surveyor General approved the plans in September 2012. Some zones were excluded from the Nature Reserve application, namely the residential sites and staff quarters in the south western corner of the farm (Figure 4).

**Advertising and Proclamation**

The MTPA advertised KPNR’s intention to become a private Nature Reserve in the Beeld and City Press and the local newspaper the Highlands Panorama for a period of 60 days. A management and notarial agreement between the KPNR and the MTPA was signed by both parties in December 2012. The intent to declare Notice has been published in the Government Gazette. At the time of writing KPNR awaits the signatures of the MEC: Department of Economic Development, Environment and Tourism (DEDET) and the Chief Executive Officer (CEO) of the MTPA to finally proclaim KPNR a Private Nature Reserve, one of the first in Mpumalanga since 1994.

**DISCUSSION**

From the time of the initial discussions to proclamation, the process of acquiring Nature Reserve status has taken 3 years of work by several shareholders and MTPA officials. The threat of mining in a pristine...
conservation worthy reserve has been averted. As a proclaimed nature reserve KPNR is exempt from attracting municipal rates. As Kudu Game Ranch becomes Kudu Private Nature Reserve a new pride and ethos will grow among shareholders and management.

A 5-year management plan has been implemented which can be modified over time. This arms the management team of KPNR with a set of tasks and measurable goals. The management plan dictates that a conservation budget must be drawn up annually to meet conservation objectives in the plan. Funding an enterprise of this nature will be a challenge. While shareholder levies will provide operational and maintenance costs, it is the establishment of a separate business, run on commercial grounds that will ensure the sustainability of the reserve. This business currently operates a selective breeding program of disease free buffalo and sable antelope. Of key note is that the development of any commercial ventures must be based on sound ecological principles to provide reinvestment in the sustainability of this unique area.

Game counts and veld condition assessments will also allow management to justify the harvest of surplus game based on measurable scientific data. Trends over time can be monitored with the objective of maintaining and in some areas improving the biodiversity and ecological processes. Burning programs which are currently in place will be reviewed annually. In addition to the annual firebreak burns, mosaic burning is practiced and the effects thereof can now be better monitored by following the management plan criteria and advice from ecologists. Erosion control will be implemented with guidance from the MTPA ecologists. Finally, conservation of the area will provide long-term monitoring sites to gauge the effects of climate change on grass:woody ratios because KPNR is located in a transition zone between grassland and woodlands. Archaeologically and culturally important sites, ruins and artefacts will also be preserved for future generations. Opportunities exist for research on various fauna and flora species and KPNR invites cooperative research endeavours with academic institutions both locally and abroad.

One of the challenges that KPNR faces is the current pollution of our rivers caused by ineffective upstream sewage treatment from the local town of Lydenburg. The KPNR is working with neighbors to bring pressure to bear on local authorities to ensure the correct operation and management of upstream sewage treatment, with financial disciplines that will ensure that local infrastructure is maintained because this could have a devastating effect on the ecology of the greater area.

Good neighborliness is a further element of the KPNR ethos. This ethos aims to encourage good relationships with neighbors and the local community as a whole. Examples of these initiatives are:

- Assisting with annual fire breaks on neighbors’ lands.
- Providing instant service and assistance for any form of veld or run away fire on neighboring properties.
- Providing fence line security to control poaching.
- Removing snares/traps on neighbors’ property.
- Entering into agreements with neighboring farms to purchase a quota of lucerne crops (used by KPNR for feeding of buffalo and sable in the breeding projects) to encourage economic development.
- Providing professional assistance and advice.

In addition it is deemed of prime importance that local school children are given the opportunity to learn more about conservation and nature. To this end, KPNR aims to develop an outreach program for local schools and to encourage, support and actively participate in conservation and ecological education programs.

CONCLUSION
We should no longer rely on National Parks alone to be the guardians of our natural heritage. The future of conservation lies in the hands of private landowners, who as custodians of the land need to recognize areas of high conservation value and to strive towards maintaining and enhancing the biodiversity, ecological processes and historical and cultural heritage of their lands.

The KPNR’s biodiversity stewardship program aims to promote the exchange of ideas and initiatives across borders in association with other privately owned nature reserves. The baseline premise for these initiatives being that it is the responsibility of private landowners of rural residential estates to manage their property in a manner that makes a significant contribution towards nature conservation. It is deemed essential that privately-owned nature reserves build associations to enable the dissemination of successful ecological management processes and methods via an open forum in a similar way to that which many online forums are based. It is envisaged that KPNR will facilitate an online forum to enable the knowledge transfer for good practices and successful policies, such that all private landowners can benefit from a pooled knowledge base and contribute towards problem solving. With mentorship from MTPA and refinement
of the management plan over time, the broad objectives of KPNR are to conserve and where possible enhance the biodiversity and ecological processes, as described in the mission and objectives.

The MEC:DEDET signed the agreement on 25 March 2013. Proclamation will be published in the Government Gazette.

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LITERATURE CITED


HOW MIGHT INTERNATIONAL CONTRIBUTIONS BE MADE TOWARDS CONSERVING AFRICA’S RICH WILDLIFE HERITAGE? SOME SUGGESTIONS

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ABSTRACT: The African continent retains the world’s richest large mammal heritage, yet African countries generally have the least capacity to conserve this legacy, which is consequently shrinking. The need for wildlife conservation to be extended collaboratively, across international borders, was raised at the International Wildlife Management Congress held in Durban in July 2012. I propose 5 ways in which this might be accomplished, entailing (1) improving scientific concepts and models to better accommodate spatial and temporal variability, (2) establishing partnerships between well-endowed agencies or parks and African counterparts, (3) removing barriers to effective ownership of local wildlife by rural communities and ranchers, (4) establishing regular regional meetings bringing wildlife scientists and managers together to address common problems, and (5) promoting the establishment of transfrontier protected areas, integrating protected areas and communally occupied land for shared economic and social benefits.

KEY WORDS: Africa, community-based conservation, partnerships, professional meetings, scientific models, transfrontier conservation areas


Diminishing populations of large herbivores and their predators even within formally protected areas in Africa have become a source of mounting concern (Caro and Scholte 2007, Craigie et al. 2010). Coupled with this is the fragmentation and isolation of populations increasingly restricted to protected areas (Western et al. 2009) and the disruption of migrations (Harrington et al. 2009). The continent’s most intensively managed national park, the Kruger National Park in South Africa, has not escaped this problem, with several of the rarer antelope species threatened with extirpation (Harrington et al. 1999, Ogutu and Owen-Smith 2003, Owen-Smith et al. 2012). Is this pattern the beginning of the faunal collapse predicted as an inevitable consequence of habitat fragmentation and range compression, from principles of island biogeography (Soule et al. 1979)? A subtly spreading effect of global climate shifts degrading local habitat conditions (Erasmus et al. 2002, Travis 2003)? Or a consequence of misdirected management interventions, like excessive surface water provision (Owen-Smith and Mills 2006)? Migratory populations are especially at risk when they wander beyond protected area boundaries during part of the year, as demonstrated by the spectacular crashes of wildebeest in Botswana’s Central Kalahari region (Williamson and Mbanu 1988) and around Nairobi National Park in Kenya (Ogutu & Owen-Smith submitted). An eternal problem has emerged in acute form, underlain by the enormous value fetched by rhino horns in the Far East; escalating poaching is pushing back the recent gains in rhino numbers in Africa towards prospects of local extirpations.

Africa’s wildlife is of international concern because of its exceptional abundance and diversity of large mammals. However, African countries are generally under-resourced economically and socially in their capacity to protect this legacy, most notably with regard to elephants and rhinos (Gillson & Lindsay 2003). If wildlife conservationists elsewhere wish to contribute towards retaining abundant African wildlife, cooperatively and across international borders, how best might they do so? This question was raised in the plenary session of the 4th International Wildlife Management Congress held in Durban, South Africa, in July 2012. Here are my thoughts and suggestions, prompted by this engagement.

The diverse constituencies committed to promoting wildlife conservation must be recognized. Most of Africa’s protected areas were established as “game reserves”, with some later elevated to the status of national parks. While hunting might be allowed in the former, it is generally excluded from the latter, with visitors thus restricted to wildlife viewing and photography. Especially in southern Africa, much wildlife is also contained in private land, managed as game ranches or amalgamated private nature reserves or conservancies. Rangelands occupied communally by people and their livestock may also retain substantial wildlife populations, potentially providing revenues through community-based resource management. Politicians may perceive wildlife-based tourism as a promising generator of new jobs and economic revenues. However, burgeoning numbers of elephants within and around protected areas have led to damage to crops threatening human livelihoods, plus breakage of fences, water pipes and other structures (Hoare 1999).
I identify 5 challenges in conserving African wildlife that could benefit from international engagement: (1) improving scientific concepts and models; (2) funding the remedial actions needed; (3) benefiting local stake-holders; (4) increasing the capacity of agency staff; (5) establishing appropriate land-use structures. I will address each of these in turn.

### BETTER SCIENCE AND MODELS

Good science can guide effective conservation action. However, the concepts and models that are commonly invoked are widely recognized as unreliable (Krebs 1995, White 2000, Norris 2004). Logistic models and various modifications thereof assume that density feedbacks restrict rates of population growth as abundance tends towards some carrying capacity. However, in many cases the density dependence of population growth may not be evident until quite high population densities have been attained, as I learnt from my doctoral study of white rhinos (Owen-Smith 1981) and others have found with respect to elephants (Gough and Kerle 2006). Moreover, it is widely recognized that “this thing called carrying capacity” (Caughley 1979) is widely variable both over space and through time, and subject to manifold influences. Moreover, the effects of crowding on resource access, exposure to predators and susceptibility to diseases differ among members of the population, being most acute for very young animals while hardly affecting the survival of prime-aged animals (Gaillard et al. 1998). Through being selective for age within the adult segment, the impact of cursorial predators on population trends can be very different from that of hunters targeting animals of both sexes (Vucetich et al. 2005). Population growth depends fundamentally on rates of resource acquisition, affecting growth in individual size and hence biomass, as well as vulnerability to predation, but models linking predator or herbivore dynamics to prey or plant populations are highly simplistic and difficult to parameterize (Caughley and Lawton 1981).

The basic shortcoming in all of the standard population models is their failure to accommodate environmental variation in time and space in a meaningful way (Owen-Smith 2011). The abundance level at which population growth rate becomes zero, defining carrying capacity, is dependent on seasonal and local variation in resource supplies and the effects of predation mortality, operating both additively and interactively with body condition and hence resource gains. These considerations are especially relevant for African savanna ecosystems harboring most of Africa’s large mammals, characterized by erratic rainfall, high spatial heterogeneity and an abundance of large mammals. The standard models need to be challenged with data and upgraded accordingly, rather than being applied in fossilized form. This is an endeavor to which all wildlife scientists should contribute. I have proposed how concepts and models might be advanced, based on my experience (Owen-Smith 2002, 2011). These “metaphysiological” (or biomass-based) models link population change directly to resource gains and the consequences for risks of predation and physiological costs. Rather than being specified arbitrarily, “carrying capacity” becomes a variable dependent on how these influences operate, specifically in regard to seasonal variation.

The task of challenging and improving these models requires long term data on population changes over prolonged periods. Much of the information derived from monitoring surveys becomes buried in archives and hence unavailable for scrutiny. There is a need to develop the cross-disciplinary field of Eco-Informatics (Peters 2010, Michener & Jones 2012), entailing not merely data gathering and storage, but also (a) the management and dissemination of these data, (b) statistical assessment of their information content, (c) modeling the understanding gained by these data, and (d) using findings to support decision-making. Only exceptionally are all of these steps followed through.

### FINANCIAL SUPPORT

Conservation agencies in African countries are chronically under-funded and under-staffed. Even the Kruger Park has abandoned the annual total counts of all large ungulates that were conducted between 1977 and 1995 (Ogutu & Owen-Smith 2003). While visitor numbers enable the Kruger Park to operate profitably, it must subsidize other national parks in circumstances where government subvention is being progressively reduced. Few agencies elsewhere in Africa attempt such surveys, and where done the findings are seldom assessed critically. Tourism revenues and jobs created become the main performance indicators, rather than the abundance and diversity of wildlife attracting these tourists. Park fees are commandeered towards meeting national development needs, rather than being fed back into wildlife conservation. This can hardly be faulted, given national priorities to uplift citizens from poverty.

Meanwhile, conservation agencies in Europe and North America have a wealth of resources, in equipment, staff and operating budgets, directed towards conserving relatively few species. This disparity has been recognized through cross-funding initiatives from which some South American countries have benefitted. No such scheme seems to operate effectively in Africa. My suggestion for overcoming the funding barrier is to establish partnerships between well-endowed national parks (or conservation agencies) in wealthier countries and smaller organizations in Africa.
and counterparts in Africa, like what has been done for cities. This could enable cross-subsidization of operating costs as well as sharing of staff, equipment and expertise. Potential benefits could accrue to both partners.

**BENEFITTING LOCAL STAKE-HOLDERS**

People living alongside protected areas containing large mammals experience the costs of crop damage and killing of livestock, but generally receive few direct benefits from wildlife. Indirect benefits may arise through the job opportunities opened and the skills thereby developed (Hackel 1999, Torquebiau & Taylor 2009). However, most of the financial benefits from wildlife accrue to government revenues and travel companies located remotely in cities. Hence the potential custodians of the wildlife resource have little incentive to conserve it.

The failure of local communities to effectively capture material benefits from wildlife means that wild animals become increasingly confined to island reserves. The crucial step towards resolving this issue is effective ownership of wildlife outside protected areas, because only when ownership is conferred will uses of this resource be promoted in such a way as to be sustainable (Owen-Smith 2012). This is clearly illustrated by failures of the CAMPFIRE program launched in Zimbabwe, because financial benefits became dissipated at district council level, rather than accruing directly to the local people bearing the costs. A striking example of success is demonstrated by Namibia. By passing effective ownership of local wildlife resources to conservancies designated by local communities, subject to democratic controls, these people gained a direct stake in ensuring that the benefits were sustained. This ownership confers the right to exploit wildlife in whatever way seems most profitable – tourism, hunting, cropping, or marketing other products. The result has been the end of previously escalating rhino poaching, and steadily growing numbers of wild ungulates and even predators outside of state protected areas (http://www.met.gov.na/programmes/cbnrm/cbnrmHome.htm). Local exploitation of wildlife can be cramped not only by national legislation, but also by international trade regulations formulated under the Convention on International Trade in Endangered Species (CITES). Export barriers have not succeeded in containing the escalating illegal trade in ivory and rhino horn. Restricting the trade to legitimately marketed products would seem more likely to be effective than completely blocking it, and thereby escalating the value of rhino horn into the realm of drug cartels.

Also relevant here are the private owners of game ranches and game farms. Their overall net contribution to wildlife conservation can be questioned in the context of the “canned” lion hunts and dubious permits for export of rhino horns as “hunting trophies”. However, the contribution that farmed rhinos might make in undercutting the illegal trade in horns, thereby alleviating the poaching pressure on park populations, cannot be discounted. Moreover, success in breeding rarer antelope species like sable and roan antelope means that these species are not endangered nationally, despite their population shrinkage in the wild in South Africa.

Protected areas alone are inadequate to secure sufficient wildlife. Those wanting to make wildlife conservation more widely effective should lobby both nationally and internationally for more appropriate regulation of legitimate uses of wildlife outside formally protected areas.

**UPLIFTING PROFESSIONAL CAPACITY**

Wildlife agencies in African countries have an increasing complement of well-qualified scientific and management staff, many with degrees from leading universities. However, their functional capacity is restricted by the financial and institutional constraints under which they operate professionally. No conference brings these scientists and managers together to address regional problems on a regular schedule, as happens in North America and Europe. This means that scientists operate largely in isolation from conceptual and technical developments in their field. Even if such meetings were to be arranged, financial resources to cover the costs of travel and accommodation would generally block attendance by most. Few African representatives from outside South Africa attended the 2012 Wildlife Management Congress in Durban, despite its venue being on their continent.

The Southern African Wildlife Management Association holds regular annual meetings, but in practice few delegates from outside South Africa attend, because of the barrier posed by travel costs. Kruger’s annual “Savanna Science Networking Meeting” has become a strong draw card for both local and international participation, but intrinsically has a strong local focus.

The lack of academic and managerial exchanges handicaps effective conservation in circumstances where knowledge and understanding are changing, new challenges emerging, and new technologies becoming available. For example, while Kruger Park managers have been closing water points to alleviate adverse consequences for large mammal diversity, in other countries new water points are being established with the short-sighted hope of counteracting local elephant impacts (Owen-Smith 1996).
What is needed are regional African wildlife management conferences held every 2-3 years, rotated among various countries. The initial region could be the countries integrated under SADC, extending from South Africa to Tanzania, although Kenya and Uganda should certainly be tagged on. These countries contain the greatest share of Africa’s wildlife wealth. This organizational structure would allow wildlife scientists and managers to meet at regular intervals, joined by international scientists wanting to contribute towards conservation in Africa.

Obviously, there must be an institutional structure to take responsibility for organizing such meetings. I suggested a regional focus within SADC, because science promoting initiatives within these countries might be conducive to supporting such meetings. The alternative would be to establish an African affiliate of The Wildlife Society, just as the International Society for Conservation Biology has regional Chapters. However, for local affiliates to be effective some funding must be passed from the wealthy parent body towards the affiliate, most importantly to cover the travel costs of attendance by African colleagues.

APPROPRIATE LAND-USE STRUCTURES

The major threat to wildlife conservation comes from the fragmentation and isolation of protected areas within an unfavorable and hazardous landscape matrix dominated by humans (Ogutu et al. 2011). The vast majority of Africa’s protected areas are too small to sustain the large mammal diversity they currently contain in the long term in isolation. The widespread wildlife declines that have been documented across the continent are symptomatic of this (Caro and Scholte 2007, Western et al. 2009). Even Kruger Park’s 20,000km² area cramps the movement patterns of elephants (Whitfield 2001), and is too narrow to provide a secure core against rhino poaching. Vegetation changes are apparent in the vast Serengeti ecosystem, with unknown consequences for its large mammal diversity (Sinclair et al. 2007). The long-term consequences for biodiversity of burgeoning elephant populations within southern African parks and surrounding areas remain contentious (Kerley et al. 2008).

The most far-sighted initiative addressing this issue is the establishment of Transfrontier Conservation Areas (TFCA), amalgamating protected areas with intervening regional landscapes including human settlements and exploitative activities (http://www.peaceparks.org). The initial developments were simply trans-boundary parks, linking South Africa’s Kalahari Gemsbok National Park with the adjoining protected region in Botswana, and Kruger National Park with the newly proclaimed Limpopo National Park in Mozambique. The most ambitious prospect is the Kavango-Zambézi TFCA planned to cover 287,000 km² extending from Botswana and Zimbabwe through Namibia, Zambia and Angola, including a regional population of over 150,000 elephants.

The problem still to be resolved is how to effectively integrate the people living within the region into such developments. For them to tolerate the costs of wildlife and associated diseases transmitted to livestock spreading from protected areas, counteracting benefits must accrue, as noted above. The implication is that appropriate forms of resource exploitation must be allowed, and the ramifying concerns for CITES, and specifically current bans on trade in ivory and rhino horn, must be confronted. Conservation biologists need to partner with economists, sociologists and politicians in addressing such broader issues. Leading resource economist Paul Collier advanced the thesis that Africa’s rural inhabitants should purchase food grown in countries where it can be produced cost-effectively by large agro-businesses, rather than attempting to grow it locally in unfavorable circumstances (Collier 2010). The money for food purchases must come from somewhere, and what Africa has to sell is the world’s richest wildlife heritage. This already forms the basis for the ecotourism industry, but currently very little of the wealth generated filters down to local level. This raises the need for structural adjustments both nationally and internationally in world trade arrangements; lobby accordingly.

SUMMARY OVERVIEW

If you, the readers, wish to promote wildlife conservation in Africa, I suggest that you act cooperatively across international borders, as follows: As scientists, improve the reliability of our concepts, models, tools and information, and make them more relevant to the spatial and temporal complexity of African environments.

To secure funding, promote international partnerships between conservation agencies, enabling those wealthier in finances to contribute where the greatest wealth in wildlife is located.

Socio-politically, promote structures that enable local people to benefit from their local wildlife resources via effective ownership.

Collegially, contribute towards the establishment of a professional body that could host regular meetings addressing wildlife conservation issues within a regional African context.

Internationally, encourage further development of TFCA and economic contexts that will make them socio-politically sustainable.
ACKNOWLEDGMENTS
I thank The Wildlife Society for the invitation to give this talk at the Durban congress, and hence present the proposals contained within it.

LITERATURE CITED


EVALUATING STRATEGIES TO FAVOR COMMUNITY PARTICIPATION IN THE CONSERVATION OF ANDEAN CATS


ABSTRACT: Community participation is a fundamental component of wildlife conservation strategies worldwide. Many tools have been employed to increase the engagement of communities in conservation programs but their results are rarely assessed. Using a case study approach from the High Andes of northwestern Argentina, we evaluate the community participation strategies adopted to support the conservation of the endangered Andean cat (Leopardus jacobita). Initially, we created a network of local Environmental Education Officers and Wildlife Monitors, providing basic training and a small stipend. Despite follow-up and networking opportunities, the majority of the local villagers we trained left the program. Subsequently, based on local people’s interest in the topic, we focused on building community capacity to start eco-tourism activities. We offered a course on environmental implications and potential of tourism, as well as a stakeholder workshop to coordinate different community-based eco-tourism initiatives and create a network to improve communication and cooperation. In spite of our initial apparent success, the network’s sustainability appeared unlikely. We suggest that our efforts produced only a minor increase in the direct commitment of communities in conservation-friendly initiatives. This illustrates the importance of evaluating community participation strategies in order to understand how they can be refined to maximize their conservation effectiveness.

KEY WORDS: Andean cat, Argentina, carnivores, community participation, conservation, eco-tourism, evaluation, High Andes, Leopardus jacobita.


In the last decades, community participation has become a fundamental component in the strategies for the conservation of endangered wildlife worldwide (Western et al. 1994). These community-based conservation actions have employed a variety of tools, from community outreach to integrated conservation and development projects (Brooks et al. 2006, Waylen et al. 2010). The increasing centrality of the human dimension of conservation requires an evidence-based approach and the development of effective, cost-efficient strategies based on the assessment of community participation measures (Sutherland et al. 2004, Macdonald et al. 2010). Nevertheless, to date, there has been little systematic investigation of the effect on conservation success of community-based approaches (e.g., Thompson and Hoffman 2003, Possingham 2012).

The Andean cat (Leopardus jacobita) is one of the most endangered felids in Latin America and worldwide (Nowell and Jackson 1996, Nowell 2002). It is considered a habitat specialist, whose distribution is essentially associated with the rocky formations found in the High Andes and northwestern Patagonia from central Peru to central Argentina (Marino et al. 2011). Its rareness and its strong habitat association make this felid suitable as a focal species for the conservation of the ecosystems that it inhabits (Marino et al. 2010). Andean cat populations are threatened by different human-related factors, including direct persecution, local elimination of major prey species, and habitat alteration (Villalba et al. 2004). Thus, the strategies for the conservation of this cat have focused on environmental education and community participation (Lucherini et al. 2012). With this goal, in 2000, we started the EduGat Program, which implemented a series of activities specifically devoted to the creation of a favorable environment for developing conservation actions in northern Argentina.

We examine here the community participation strategies adopted by the EduGat Program to support
conservation of the Andean cat in the High Andes of north-western Argentina, with the objective of providing a preliminary, qualitative assessment of their effectiveness.

**STUDY AREA**

Community participation activities were carried out in the High Andes region of the north-western Argentina province of Jujuy. This is a remote and sparsely inhabited area, located at an approximate elevation of 3,500-4,000 m. Livestock (mostly llamas) are the primary source of income for the small (20—150 families) rural communities of Coranzuli, Cusi Cusi, Lagunillas del Farallón, and Loma Blanca, where we worked.

**METHODS**

We adopted 2 main strategies to engage communities in conservation. Initially, we aimed at creating a network of local Environmental Education Officers (EO) and Wildlife Monitors (WM), by providing their basic training and a small stipend. Our major expectation was that EO and WM would act as multipliers of our conservation message. Education Officer capacity building lasted 12-24 hours (depending on the level of trainees’ previous knowledge) that included background information on general ecological concepts and a specific, practical training in environmental education tools and strategies. We provided all EO with a copy of an Environmental Education guidebook that contained a range of activities we designed specifically for High Andes primary school students. Afterwards, we made a great effort to keep in contact with all EO as frequently as possible and offered them access to additional information and education materials as well as the possibility of participating in networking activities. Finally, we provided a small economic incentive that was directly proportional to the number of hours that EO devoted to environmental education activities. We selected WM among the local people who were hired to help with our project’s field data collection campaigns based on their motivation level. The WM training included practical, field-based experience on GPS and camera trap handling, as well as data sheet filling combined with theoretical, informal talks on carnivore ecology and their study mostly focused at clarifying the use of the data collected. Subsequently, we asked them to set up and regularly monitor a few camera traps. We visited them 2—3 times per year to collect the information gathered and to pay a small stipend proportional to the time they had devoted to field work.

Since 2010, based on local interest in the topic and in cooperation with an important native women’s association, our second strategy has focused on building community capacity to develop eco-tourism activities by (1) delivering specific information to local villagers that would increase their understanding of eco-tourism activities, and (2) creating a network of organizations and institutions that would support the implementation of community-based, environmentally-friendly tourism initiatives. To accomplish these objectives, we organized a participatory course on both the environmental implications and development potential of community-based eco-tourism tailored to the specific characteristics of the High Andes environment. The 3- to 4-hour-long course was then given in 3 villages in the form of talks open to the whole community. At that same time, we organized a stakeholder workshop with the theme “Towards the development of sustainable tourism in the Highlands of Jujuy” (Abra Pampa, Jujuy province, April 2011). In addition to representatives from the local communities, we invited all agencies from provincial and national governments, and NGOs with a connection to eco-tourism that we were able to identify. When necessary, we offered to cover the delegates’ expenses to attend the workshop.

Evaluating the success of conservation programs is not an easy task (Kleiman et al. 2000). No single perfect technique exists, and relying exclusively on a final evaluation of results may not provide adequate feedback for the improvement of strategies (Jacobson et al. 2006). Because we were primarily interested in the improvement of the conservation status of a given species, the best quantitative measure of success was to monitor the population trends of the species we were trying to conserve (i.e., ecological success; Brooks et al. 2006). Nevertheless, in the case of rare and elusive wildlife, such as the Andean cat (Marino et al. 2010) this was impractical. A second, equally impractical measure of success was to test whether our actions had obtained a positive change in the behaviour of people towards the desired ecological outcomes (Brooks et al. 2006). Because of time and resource constraints, and because the EduGat Program was implemented to respond to a conservation emergency, we selected a less immediate, but more applicable tool to assess the effects of our strategies: we monitored local levels of participations in our conservation-related activities. The assumption underlying this approach was that the level of community participation in this type of activity is positively related to their commitment to conservation. Whenever possible, we also attempted to detect the reasons behind decisions to participate.

**RESULTS**

We trained 8 rural school teachers and 6 young local villagers as EO, and 8 local villagers as WM from 2006 to 2010. At the end of 2011, only 2 EO (14.3%)
and 3 WM (37.5%) were still active. Thus, the overall desertion rate was 77.3%. In the majority of the cases where we were able to get relevant information, trainees left our project because they either moved away from the region in search of less arduous employment conditions (especially teachers) or they found more remunerative jobs. Some teachers who did not move away simply appeared to have lost interest. The courses on eco-tourism delivered to local communities were attended by a total of 60 adults (16-55 years of age), corresponding approximately to 12-17% of the residents of the 3 villages.

The 2-day-long workshop we held at Abra Pampa was attended by 21 delegates, representing 3 local communities, 3 NGOs, 1 native women’s association, 2 provincial governments, and 3 national government institutions. Three NGOs and 1 national governmental agency also invited did not attend. The workshop was instrumental in analyzing the difficulties and problems experienced by attendants while planning and implementing sustainable tourism, and it identified the potential risks threatening its future development as an acceptable source of alternative incomes for local communities of the High Andes. The most relevant actions that were needed to improve the proposals and projects for community-based eco-tourism in the region were then identified. Finally, participants discussed how each organization or institution could cooperate to develop and implement a sustainable tourism program for the Highlands of Jujuy, identifying specific contributions. Although all the representatives agreed to the need to form a network with the goal of favoring the development of such a program in the region, until now, none have taken any further action.

**DISCUSSION**

Ideally an evaluation of behavioral changes would demonstrate whether our project’s initiatives to increase community participation had its intended effect. However, it is clear that the conservation-support network we aimed to create met with limited success. The analysis of these factors would require specific research that is beyond the scope of this preliminary assessment; however, we have ideas about the causes of this limited success. Two major objectives of our strategies were the implementation of a participatory approach to conservation and the provision of alternative, conservation-related sources of income to local people, which were expected to act as an incentive for pro-conservation behaviors (Larson et al. 1998). Given the fact that the strong linkages between development and conservation are increasingly recognized, modifying community attitudes is a long and difficult process, which may require strategies favoring community participation, as well as a regional plan for economic development (Western and Wright 1994, Romero et al. 2012). Thus, it is probable that the economic incentives we were able to provide to EO and WM were too low. The great majority of the rural school teachers that we trained initially moved to less-remote schools as soon as they had the chance. Although this problem could be overcome by our later decision to concentrate training effort mainly on local villagers, there was no evidence that this new approach was successful, because many of these EO and many WM, also decided to move away or take different jobs. This was probably related to the fact that, during most of the study period, there was a large active mining operation in the region that hired a large proportion of the local labor available by offering much higher salaries.

In the case of our eco-tourism related initiatives, it is possible that local people lost interest because they failed to see short-term benefits. However, we argue that the lack of interest by governmental authorities and passive attitudes by local people are 2 additional important factors. The latter may be due to a complex set of social, political, and historical factors, which have created a culture of dependency on external or government support, an issue that may be addressed by a specifically-designed community participation strategy.

Most community participation initiatives are expected to indirectly increase tolerance by increasing awareness. Moreover, there is anecdotal evidence that the large environmental education component of our project, linked to the community engagement activities described here, have improved local awareness of Andean cat conservation within the population of our study area. Nonetheless, we conclude that our efforts produced only a minor increase in the direct commitment of communities to conservation-friendly initiatives. Thus, in spite of all its methodological limitations, this case study illustrates the importance of evaluating community participation strategies to understand how they can be improved to maximize their conservation effectiveness. Furthermore, the results of this evaluation provide support for the arguments that successful initiatives to engage local communities in conservation, require that the underlying political, economic, social, and cultural context that shapes people’s behaviors be not only understood but also influenced (Macdonald et al. 2010, Waylen et al. 2010). In our experience, this can be achieved only through a long-term, constant presence and commitment with the local communities.

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LITERATURE CITED


MITIGATION TO MINIMIZE MORTALITY ALONG THE ALL-AMERICAN CANAL, CALIFORNIA, USA

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ABSTRACT: Colorado River water is transported into California via the All-American Canal to support agriculture. From 2007—2009, a 37-km portion of the canal was lined with cement to prevent water seepage into the ground. When lined canals are constructed in deer (Odocoileus spp.) habitat, deer falling in canals and drowning contributes to mortality. From 2004—2012 we monitored mule deer (Odocoileus hemionus) use of the All-American Canal, Imperial County, California. Our objective was to monitor use of the All-American Canal by deer using track plots, road and aerial surveys, and to monitor the number of deer that died in the All-American Canal. Prior to lining the canal, mule deer were reported near the canal during 2001—2004 and there was a single deer drowning incident. During the lining and post-lining phases of the canal, 5 mule deer became trapped in the canal, 2 deer drowned, and 3 were rescued and released, despite regular use of the All-American Canal to obtain water during the hottest part of the year. In contrast there have been >550 human deaths in the All-American Canal since 1942. During the lining phase of the canal the only safety measures implemented for humans were safety ladders. It was not until a national network news organization reported on the dangers of the All-American Canal that the California Legislature authorized funding for the lining project. The All-American Canal was lined with cement in 2007—2009 (U.S. Bureau of Reclamation 2006).

Canals act as attractants for ungulates by providing adjacent forage (Michny and McKevitt 1982, Gatz et al. 1984), cover (Gatz et al. 1984), and water (Krausman and Hervert 1984, Rautenstrauch 1987). However, canals act as barriers to movement to and from foraging areas, winter and summer range, and dispersal areas for ungulates (Shult 1968, Menzel 1969, Michny and McKevitt 1982, Busch et al. 1984, Fry et al. 1984). The most serious consequence of cement-lined canals for ungulates is the increased risk of drowning. Numerous studies have documented ungulates becoming trapped in cement lined canals resulting in death (Boulders and Bailey 1980, Busch et al. 1984, Krausman and Hervert 1984, Krausman 1985).

The Coachella Canal, a north-running branch off the All-American Canal, was responsible for the drowning deaths of 29 mule deer during the construction phase (i.e., cement-lining) in 1980 and 18 in the first year post-lining (Rorabaugh and Garcia 1983). Approximately 200 mule deer have drowned in the Coachella Canal after it was lined with cement (L. Lesicka, Desert Unlimited, unpublished data). Large mammal (e.g., mule deer and bighorn sheep [Ovis can-
The climate of the Sonoran Desert of southeastern California is arid with rarely freezing temperatures in the winter and temperatures frequently >45°C in the summer. Annual precipitation averages 74 mm but is highly variable (range 4—216 mm; Imperial Irrigation District, unpublished data). Vegetation is typical of the Lower Colorado River Valley Subdivision of Sonoran desert scrub (Brown 1994). There are 4 main vegetation associations: creosote bush (Larrea tridentata), scrub, psammophytic scrub, microphyll woodland, and canal-influenced vegetation. Creosote bush scrub is the most common and typical plant species include creosote bush, brittlebush (Encelia farinosa), and burrobush (Ambrosia dumosa). Psammophytic scrub occurs in the dune system and the vegetation is adapted to shifting sand. Typical vegetation includes Nevada Mormon tea (Ephedra nevadensis), Colorado Desert buckwheat (Eriogonum deserticola), desert twinbugs (Dicoria canescens), sandpaper plant (Petalonyx thurberi), desert panicgrass (Panicum urvilleanum), and fanleaf crinklemat (Tiquilia plicata). It also includes the rare or endangered Peirson’s milk vetch (Astragalus magdalena var. peirsonii), Algodones sunflower (Helianthus niveus tephrodes), Wiggins’ croton (Croton wigginsii), giant Spanish needles (Eriogonum deserticola), and Borrego desert scrub (Borrego plicata). The microphyll woodland is the alluvial fan dissected by drainages from the Chocolate and Cargo Mucha mountains. Typical vegetation include blue paloverde (Parkinsonia florida), desert ironwood (Olneya tesota), and smoketree (Psorothamnus spinosus). Canal influenced-vegetation include cattails (Typha spp.), spotted ladythum (Persicaria maculosa), horseweed (Conyza canadensis), spiny chloracantha (Chloracantha spinosa), giant reed (Arundo donax), small-flowered tamarisk (Tamarix parviflora), false daisy (Eclipta prostrata), common sunflower (Helianthus annus), white sweet-clover (Melilotus albus), and arrow weed (Pluchea sericea) (RECON Environmental 2009).

METHODS
We monitored use of the All-American Canal (from Pilot Knob 38 km west to Drop 3) 2004-present. The canal was monitored in 3 phases of construction. The pre-lining phase was from November 2004 through 2006. The lining phase began in 2007 through 2009. The post-lining phase was from 2010 to present. During the pre-lining phase, we surveyed the project area for their use a combination of track plots (Popowski and Krausman 2002), road surveys for sign (i.e., tracks, pellets, observations of deer), and aerial surveys from a Cessna 172 (Krausman and Etchberger 1993). Road and track plot surveys were conducted weekly and aer-
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Trial surveys were conducted monthly with a pilot and 2 observers during the pre-lining and lining phases of the canal. During the post-lining phase of the canal we conducted bimonthly road surveys from May through November. We also placed an infrared motion detection camera at the canal to determine the rate of use by mule deer of the canal during the post-lining phase. Most observations and plots were established south of the All-American Canal near Pilot Knob; however, we also surveyed the area north of Interstate 8 (I-8) at the junction of the canal and I-8 west of Pilot Knob.

During the study, we also became aware of the number of humans that drowned in the canal since construction. Thus, we discuss those deaths and mitigations compared to the mitigation for deer.

RESULTS
We detected no deer or deer sign south of the All-American Canal. From 2004 to 2006 we did not detect deer using the canal. We first encountered deer coming to the All-American Canal during summer 2007. All instances of deer coming to the canal occurred at the area north of I-8 at the junction of the canal and I-8 west of Pilot Knob. Use of the All-American Canal was highest during the hottest part of the year (Figure 1). We encountered mule deer at the canal on 2 occasions: 1 adult male in August 2007 and 1 adult female in July 2008. Prior to the completion of construction we noted scrape marks in the mud and algae on the cement sides of the canal where deer had fallen in and attempted to exit the canal but were not able to gain footing on the steep cement sides. These deer were able to float down to the unlined portion of the canal where they could safely exit as evidenced by deer tracks in the sand. Adult males, females with fawns, and yearlings visited the canal during the evening hours nearly every day in summer. We were not able to determine how many deer were visiting the canal or whether they were the same deer visiting the canal repeatedly.

Overall we detected 5 mule deer that had become trapped in the All-American Canal. Two of the 5 deer drowned. The first incident occurred in November 2009 when an adult female drowned during the lining phase of the canal. The second incident occurred in January 2011 when an adult male drowned post-construction. The 3 other deer that were trapped in the canal were rescued and released. The first incident involved 2 fawns that were rescued by the construction crew in November 2009. The second incident occurred post-construction in October 2011 involving an adult female that was rescued and released by the Imperial Irrigation District and the California Department of Fish and Game.

Since the All-American Canal was constructed, >550 humans (approximately 8 people/year) drowned in the canal, likely moving into the U.S. from Mexico illegally. During 2007—2010, 42 people drowned in the study area.

DISCUSSION
In comparison to the Coachella Canal, very few deer drowned in the All-American Canal. Sand dunes are not quality habitat for mule deer; however, the canal acts as an attractant causing mule deer to move through sand dunes to obtain water (Marshal et al. 2006a). Marshal et al. (2006b) estimated the population home range of 87 km$^2$, well within the distance traveled by mule deer to the All-American Canal. Expansion joints (approximately 2-m unlined segments covered with rock) built into the canal act as a safe point of access/egress to the canal for deer. We believe these small areas have prevented higher drowning rates. We do not have estimates of the number of deer visiting the canal for water. Marshal et al. (2006b) estimated a population size of 40—106 mule deer during his 5-year study. It would also be interesting to know how mule deer are using the sand dunes, and whether they are only moving through them to the canal or using them in another capacity.

The high number of mule deer drownings in the Coachella Canal united a number of stakeholders (e.g., Desert Unlimited, California Department of Fish and Game, Imperial Irrigation District, U.S. Bureau of Rec-
lamination) to prevent a similar situation from occurring during the lining and post-lining of the All-American Canal. Experts were brought into survey the area because of reports of deer near the canal on the Quechan Indian Reservation during 2001—2004. Nevertheless, it was assumed deer did not move through the sand dunes to the canal because of lack of forage (Imperial Irrigation District 2004). Mitigation measures were also discussed prior to the All-American Canal lining. Experimental large mammal escape ridges had been tested on a portion of the Coachella Canal to allow safe passage for mule deer and bighorn sheep. The large mammal escapes ridges proved to be structurally unsound, weakening the concrete lining causing seepage. Implementing large mammal escape ridges was removed from construction plans for the All-American Canal and off-site mitigation was suggested pending the outcome of deer surveys (U.S. Bureau of Reclamation 2006). In 2009 the Imperial Irrigation District stated deer near the All-American Canal warranted off-site mitigation rather than the expense of completely fencing the area to prevent deer access to the canal. Two wildlife guzzlers were placed north of the canal to prevent mule deer from moving south to obtain water (U.S. Bureau of Land Management 2009). These wildlife guzzlers were additions to 21 other wildlife guzzlers, and have not prevented deer visiting the canal to obtain water.

There have only been 3 reported deer that have drowned in the All-American Canal. During the course of the study, however, we learned that there were >550 humans that have drowned in the All-American Canal. Approximately 11.5 million undocumented immigrants resided in the U.S. in 2011 and 2.8 million resided in California, the state with the highest undocumented immigrant population. About 14% entered the U.S. later than 2005. Fifty-nine percent of undocumented immigrants in the U.S. are from Mexico (Hoefer et al. 2012). The U.S.—Mexican border in Imperial Valley, prior to the construction of the planned 3,169 km border fence in 2008, offered a single obstacle for migrants entering illegally, the All-American Canal. Strong undercurrents of the canal create a drowning hazard for people attempting to cross (Ellingwood 2004). Prior to lining the All-American Canal approximately 500 people drowned in the canal from 1943—2006, more than half were Mexican citizens. During 1980—1990, 143 undocumented immigrants drowned in the All-American Canal (53% of drowning deaths in the canal; Agocs et al. 1994). In 1997, 1998 and 2001, 43, 95 and 91 drowning deaths occurred in the canal respectively (Hinkes 2008). Peak drowning deaths occurred in 1998 (Corneilus 2001, Eschbach et al. 2001).

In 2008 the Imperial Irrigation District wrote a letter to Fern Steiner, president of the San Diego Water Authority (the agency responsible for funding the lining project) expressing the desire to do “everything that can be reasonably done to minimize the safety hazards...on purely humanitarian grounds” regardless of the increased costs. Measures included safety ladders, buoy lines, and bilingual signs placed along the canal to help someone stranded to exit safely (http://www.allamericanканal.org/). Safety ladders, spaced 114 m apart, were the only safety measure implemented during the lining-phase of the All-American Canal (U.S. Bureau of Reclamation 2006).

Forty-two people have drowned in the All-American Canal during 2007-2010 (http://www.allamericanканal.org/). In May 2010 CBS News reported on the dangers of the canal, referring to it as the “the most dangerous body of water in the United States” Consequently in 2011, 105 buoy lines with orange floats and strobe lights and bilingual signs were installed every 0.8 km along the cement lined portion of the canal (Imperial Irrigation District 2011). Since the new safety measures were installed along the All-American Canal there have been no drowning deaths in canal; however, the U.S. Customs and Border Protection rescued 3 undocumented immigrants who were Mexican citizens from the canal in May 2012 (U.S. Customs and Border Protection 2012). The U.S. Customs and Border Protection stated that the decline in human mortality could be a result of a number of factors: the border fence erected in 2008, the increase in the number of border protection agents, the economic decline in the U.S., the new safety measures installed along the All-American Canal, or some or all of the above (Varin 2011).

Our objective herein was to look at the number of deer drowning in the All-American Canal but quickly became aware of a much larger problem: the number of people drowning in the All-American Canal. Numerous groups came together to prevent a repeat occurrence of the ungulate mortality in the Coachella Canal by taking measures prior to the lining phase. These deer have been afforded no special protections (i.e., endangered, threatened, or species of special concern) and have been a hunted population. Only 3 known mule deer drownings have occurred in the All-American Canal. In contrast, peak drowning deaths for humans in the All-American Canal occurred in 1998 nearly a decade before the canal was lined. The only safety measure implemented during the lining were safety ladders, which most likely would be ineffective because even strong swimmers would struggle to reach the ladders with the increased flow rate of water post-lining. On 1 occasion we observed a young, adult Mexican man traverse the unlined canal exiting nearly 1.6 km down-
stream from where he entered the canal. It took nearly 2 years and a national news story after lining the canal before further safety measures were implemented. If the All-American Canal had been located in another part of the U.S., would >550 people drowning in the canal be tolerated? And, is not a human life worthy of similar consideration shown to that of a deer? As biologists working in an area we must take into consideration the human and biological community including the humanitarian aspect. Working to give the community a stake in wildlife populations gives a larger voice to wildlife management and conservation. However, it also encompasses looking at broader issues in a community and working with experts from many different fields to lead to changes which benefit wildlife and the human community.

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INVOlVING COMMUNITIES IN WILDLIFE RANCHING IN ZIMBABWE: A GRASS-ROOTS INITIATIVE

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ABSTRACT: We present the findings of a feasibility study on Commercial Communal Wildlife Ranching (CCWR) in selected semi-arid districts of Zimbabwe. The specific objectives of the study were to: (1) learn from past and on-going experiences in the sub-region; (2) conduct targeted field studies; and (3) identify potential income generating activities which are compatible with CCWR. The methodology involved the review of official documents and published literature on game ranching in Southern Africa. An assessment matrix with 8 components aided the exploration of seven districts followed by a selection of pilot sites where participatory rural appraisal methods were used. In these areas poverty is prevalent and livelihoods, based on dryland agriculture, are precarious; however, the areas are rich in wildlife. Legislative changes in the 90s and 90s in South Africa, Namibia and Zimbabwe devolved custodial rights over wildlife to private, mostly white, landholders. Huge economic and conservation benefits have been derived from this, thus prompting current governments to extend game ranching to communal lands. As in South Africa, game ranching in Zimbabwe is mostly carried out by large scale commercial farmers — however, during the 1990s, Ward 4 in Mbire district established Chivaraizde Game Ranch using the Communal Areas Management Program for Indigenous Resources (CAMPFIRE) principles. People in study wards are keen to improve on their CAMPFIRE experiences by starting CCWR. They do however face financial, technical and institutional challenges.

KEY WORDS: CAMPFIRE, community-based management, game ranching, wildlife ranching, Zimbabwe


Commercial wildlife ranching is defined as the management of game in a sizable game fenced system, with minimal human intervention in the form of water provision, food supplementation during periods of drought, strategic control of parasites, and provision of veterinary care. It includes all forms of wildlife-based land use which can be promoted in a game ranch including hunting, live animal sales, and ecotourism. These in turn make significant contributions towards wildlife conservation and poverty alleviation if benefits are distributed equitably (Dry 2011). The principle of achieving a higher level of production from the utilization of a diverse group of species, each adapted to use a different component of the vegetation (e.g., grazers, browsers, and mixed-feeders), was formulated by ecologists studying large concentrations of game in south and east-central Africa. There is also the belief that game ranching requires a relatively low level of management expertise (Fairall 1984).

In Zimbabwe, South Africa and Namibia, wildlife-based land use on freehold farmland was established during the 1960s and 1970s following legislative changes that bestowed custodial user rights over wildlife to private landholders. The farmers involved united their properties and managed wildlife for lucrative commercial uses. Important insights with far reaching implications for communal game ranching as well as community-based natural resource management have been derived from commercial wildlife ranching. This can be summed up as follows: (1) in semi-arid and arid ecological contexts, wildlife can be a more sound and economically productive form of land use when compared with livestock production; (2) the state does not have the resources to manage wildlife everywhere and the most effective managers of wildlife are the people who live with it and pay for the costs of its existence; (3) the people who live with wildlife will only use and manage it sustainably when they have secure rights to manage and reap the full rewards of their management inputs (Murphree 1996, Child 2009).

These insights helped in the conceptualization and implementation of the world-renowned Communal
Areas Management Program for Indigenous Resources (CAMPFIRE), a program based on the hunting of free-ranging wildlife outside protected areas (Murphree 1991). With exception of Chivaraidze Game Ranch in Mbire District (LeBel et al. 2004), CAMPFIRE has largely not actively involved communities in establishing game ranches where they can intensively manage wildlife for economic and other benefits. The advent of the Wildlife Based Land Reform Policy of 2004 in Zimbabwe provides a good opportunity to extend these insights to communities living in agriculturally marginal and under-developed, but wildlife rich areas of the country. The hypothesis is that Commercial Communal Wildlife Ranching (CCWR) will not only provide the classic benefits of a commercial wildlife ranch (e.g., revenue, venison, employment) but will also include a range of services which improve the livelihoods of local communities and act as a hub for local development.

The overall objective of this paper is to explore, based on recent grass-roots initiatives, the extent to which CCWR can be an opportunity for local communities with adequate wildlife resources. After a quick overview of CAMPFIRE, the background of this initiative, we explain how local communities stimulated by the private sector are revisiting CAMPFIRE. From the description of the rational approach used to select the most promising CCWR sites, the potential of this new wildlife land use option will be demonstrated using the Mahenye CCWR initiative as an example.

**STUDY AREA**

Zimbabwe is a landlocked country in southern Africa with a total surface area of 391,000 km², much of the country is semi-arid with rainfall ranging between 400 mm and 650 mm per year (Ministry of Environment and Natural Resources Management 200). The estimated human population is about 12 million with 70% of the Zimbabwean population living in the rural areas. Poverty continues to be a great concern, especially in the Kalahari ecoregion in the northwest, the Zambezi ecoregion in the north and Save-Limpopo ecoregion in the south where rain fed crop production, the most common method of meeting livelihood

![Figure 1. Commercial communal wildlife ranching sites in Zimbabwe selected for initial evaluation.](image-url)
demands, is constrained by a low and erratic rainfall regime. Livestock farming is also an equally risky enterprise due to endemic diseases, unpredictable droughts, limited access to high quality grazing and markets (Mitchell 2001). As a result of living in such fragile agro-ecological conditions characterized by uncertainty and limited livelihood options, most households still depend largely on wild food resources and assistance from wealthier households, government and non-governmental organizations (Cunliffe 2010). To strengthen conservation and development in these impoverished but wildlife-rich communal areas, CAMPFIRE was initiated in the late 1980s by the Zimbabwean Government with the support of international donors (Murphree 1990, Taylor 2009a).

CAMPFIRE: Devolution and Local Governance
Conceptually, the philosophy behind CAMPFIRE is that local communities will sustainably manage wildlife and other resources when the rights and responsibilities to protect and use wildlife are devolved to them as managers (Martin 1986), benefits of managing the natural resources exceed the costs (Murphree 1990), communities enter into business partnerships with the private sector (Katerere 2002), the benefits of wildlife conservation are captured by the local communities as resource managers, and communities are small enough to be cohesive (Murphree 1991). Rural people will protect and sustainably use wildlife and related natural resources when the benefits of doing so are perceived to exceed the costs (Murphree 1991, Duffy 2000, Baldus 2009) and this will in turn increase tolerance towards wildlife species, especially elephants (Loxodonta africana; Taylor 1993). As Bond (2001) points out, economic benefits are considered to provide the most important motivational factor influencing local people’s interest and participation in community-based wildlife management. During the first 15 years of CAMPFIRE (1989-2006), over $30 million USD was generated for the communities of the 19 Rural District Councils (RDCs) fully participating in the program, most of this income coming from sport hunting (80-90%) and predominantly (60-65%) from elephant hunting (Taylor and Cumming 1993, Bond 1994, Maveneke 1996, Frost and Bond 2008, Taylor 2009a).

The CAMPFIRE policy initially proposed that at least 55% of gross wildlife revenue should be devolved to Ward level, up to 30% could be retained for wildlife management purposes at RDC level, and no more than 15% retained as a council levy (Taylor 2009a). The last improvement in the CAMPFIRE model was the direct payment system allowing the safari operator to pay the 55% of wildlife revenue directly into the Ward bank account (Taylor 2009b).

A Wildlife Resource at Risk
Since 2006, CAMPFIRE has been impacted by the political and economic crises prevalent throughout the country which has reduced the quality of governance and community benefits (Balint and Mashinya 2008). There has been a general feeling that the program is failing to resolve human-wildlife conflicts, particularly when operational costs of managing wildlife far exceed the benefits to local communities (Gadzirayi 2007, LeBel et al. 2011) and promised benefits are not received (Fischer et al. 2005, Rihoy et al. 2010). It emerged that household incomes from CAMPFIRE participation were often insufficient to offset crop damage. In addition, this was aggravated by the disproportionate allocation of safari revenues in favor of the RDCs (Logan and Moseley 2002, Rihoy et al. 2010). The recent decline in distribution of revenues to sub-district levels was largely due to a deterioration of the national economy in the 2000s, associated with exchange rate distortions and the lack of commitment by RDCs to disburse scarce financial resources (Taylor 2009a, Rihoy et al. 2010).

The Wildlife Based Land Reform Policy (WBLRP) of 2004
As a result of the perceived shortcomings of CAMPFIRE, some local initiatives to secure wildlife resources were implemented in various districts known to be pristine wildlife areas and these projects were supported by informal partnerships between safari operators, community-based organizations, and RDCs. These grass-roots activities aimed to establish large, fenced protected areas and were an encouraging signal to explore the possibility of promoting CCWR. The
government’s draft WBLRP of 2004 encourages such initiatives in the context of Zimbabwe’s land reform program. However, the latter emphasized crop and livestock production to the exclusion of wildlife and hence fuelled poaching (Lindsey et al. 2011), habitat degradation and woodland loss in newly settled areas (Wolmer et al. 2003). The WBLRP recognizes that wildlife is a viable land use option especially in agriculturally marginal areas. Its principal objectives are to ensure more equitable access by the majority of Zimbabweans to land, wildlife resources, and business opportunities that stem from the use of these resources and to develop and implement appropriate institutional arrangements for wildlife-based land reform.

METHODS
Assessment of Potential CCWR Sites and Participatory Rural Appraisal
The selection of potential CCWR sites emerged from private and local initiatives conducted between June 2010 and January 2011 under the auspices of Food and Agriculture Organization of the United Nations (FAO) with the support of the Zimbabwe Parks and Wildlife Management Authority (ZPWMA). Seven RDCs were involved in the north and the south of the country (Figure 1), namely Hurungwe, Mbire, Nyaminyma, Binga in the Zambezi ecoregion and Chiredzi, Chipinge, and Bikita in the Save-Limpopo ecoregion and all these ecoregions have a rich diversity of both flora and fauna.

Based on a previous study (Benson 1991), an assessment grid was developed locally to compare the different initiatives using a set of 8 variables that helped to give an overall picture of each selected project (Figure 2) including its natural potential (variables 1 to 3), partnership and economic perspectives (variables 4 and 5), its social background (variables 6 and 7) and its political context (variable 8). After a ranking exercise, only the most promising sites were visited to clarify some points with the local authority and community representatives. After meeting with district-level officials and some community representatives, participatory rural appraisal exercises were conducted in January 2012 with community participants, including traditional and elected leaders.

Identifying Income Generating Activities
During the feasibility study, attention was also placed on identifying a range of potential and compatible income generating activities in which the communities
could engage. Non-timber Forest Products (NTFP) and carbon markets were analyzed and assessed in respect to how they could potentially increase the income of CCWR project sites. For the carbon stock of the forest cover, we used the values reported from a study undertaken in a similar ecosystem (Ryan et al. 2011).

Business Plans
Anticipated business plans with estimated income and expenditures were developed with the representatives of communities, RDCs and safari operators. A specific focus investigated the background of the previous CAMPFIRE project, the proposed CCWR project, any work in progress and the extent to which each party was willing to invest.

RESULTS
CCWR Model: a Local Hub for Conservation and Development
The concept of CCWR which emerged from this exercise aims to integrate wildlife ranching with agriculture in communal land areas where most households are primarily concerned with subsistence farming. This CCWR model (Figure 3), based on a commercial partnership between a dedicated community and its private partner, should not only promote the classic wildlife-based activities but also recognize the local needs and act as a local hub providing agricultural services for the community and its neighbors.

Ranking Local Initiatives
From the 7 targeted districts, the Nyaminyami RDC initiative was removed from the study as they failed to meet the criteria in the evaluation grid. From the remaining 6 districts, 11 CCWR proposals totalling 1,150 km² were analyzed with the in-house evaluation grid and the results of the assessment synthesized in spider diagrams (Figure 4). Initially, proposals with poor wildlife resources or inadequate size (less than 5,000 ha) were rejected; this was the case with the proposals from Binga and Hurungwe RDCs. Projects involved in political wrangling or social instability were withdrawn, as was the case with Bikita, where district officials prevented us from conducting field work. Projects with poor business plans, unclear partnership arrangements or poor community involvement were rejected; this was the case for Mbire and Chiredzi RDCs. Thus, the remaining project was the proposal from Mahenye in Chipinge RDC.

Communities’ Expectations and Development Constraints
Besides the wildlife resource base for both consumptive and non-consumptive uses, the strong political support from most of the RDCs, local leadership and the existence of local wildlife management institutions, several key challenges emerged from the feasibility study.

Lack of alternative livelihood options. –Subsistence agriculture and limited livestock husbandry are the two main livelihood options in the study areas. With the exception of Chivaraidze Game Ranch in Mbire District (LeBel et al. 2004), the involvement of local people in economic activities based on communal game ranching was previously non-existent.

Shortcomings in institutional framework.—The Parks and Wildlife Act devolved wildlife use rights to RDCs. The policy makers hoped that RDCs would devolve these rights further to ward-level producer communities but this has not taken place. This has become an enduring source of tension between RDCs and the producer communities who bear the costs of living with wildlife. Additionally, there is a need for greater clarity with regard to benefit sharing between RDC, safari operators and local communities.

Wildlife management and household poverty alleviation.—To date the distribution of CAMPFIRE
revenues has a nebulous link to household poverty alleviation, gender mainstreaming, and HIV/AIDS. The impact of CAMPFIRE has not been sufficient to alleviate poverty which continues to persist and in some areas is actually deepening. The focus has mainly been on community development and this has taken the form of community projects such as grinding mills, purchase and operation of tractors and trucks, and construction of schools and clinics. Except for 2 producer wards, the direct payment system of CAMPFIRE revenues is not implemented.

**Human-wildlife conflicts.** Additionally, wildlife is viewed as a challenge to livelihood security because of crop damage, attacks on livestock, and the danger posed to human life. The ZPWMA is perceived as placing a greater value on protecting wild animals rather than enhancing the basic quality of human life and this generates resentment. Greater numbers of wildlife continue to be lost to poaching while some wildlife habitats are converted to crop agriculture.

**Lack of access to NTFP markets.** At all sites, there is no vibrant trade of forest products as all community members access these resources for subsistence use and as a coping mechanism in times of extreme food stress, particularly during January to March when there are acute food shortages. Thatch grass is commonly traded between households but at a very low level and is occasionally sold at $0.50 USD per bundle. Other than this poorly defined trade, the forest products are used for subsistence use and they are not sold inside or outside the wards.

**Lack of access to forest carbon markets.** The study communities have maintained good forests, but lack access to markets for vital forest environmental services including carbon sequestration. The calculation of the current carbon stock of the CCWR sites was evaluated at 86tC/ha (77% from forest and 33% from grassland) which should allow them to earn carbon credits from the voluntary market through avoided deforestation strategies.

### The Mahenye Case Study

The most robust proposal of Mahenye in Chipinge RDC was selected as a pilot project to explore how game ranching could contribute to lasting changes in community livelihoods and the sustainable management and use of wildlife and other natural resources. Bordering Mozambique and on the edge of Gonarezhou National Park the initiative concerns Ward n°30 with 960 households and aims at creating a CCWR of 100 km² (Figure 5).

**Background.** Wildlife use of this landscape is variable, during the rainy season populations are high but during the dry season there is migration back into Gonarezhou National Park and into Mozambique due to the lack of surface water. This has resulted in low quota utilization by the safari hunters, limited benefits for the communities and a lot of mistrust between the communities, and the safari operators or the RDC.

**The proposed CCWR project** – A Bonnox™ fence, impossible to use as snares, will be used to fence off the boundary of the CCWR, boreholes will be drilled, and a restocking program of about 500 game animals will be initiated. Game scouts will be equipped, employed, and monitored to protect the wildlife capital.

### Table 3. Provisional annual budget (in USD) of the Mahenye commercial communal wildlife ranching project.

<table>
<thead>
<tr>
<th>Income</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Sale of Meat, 10 metric tons</td>
<td>$20,000</td>
</tr>
<tr>
<td>Hunting, 5 elephant hunts</td>
<td>$150,000</td>
</tr>
<tr>
<td>Tourism, 3,000 guests</td>
<td>$60,000</td>
</tr>
<tr>
<td>Sundry, Sale of craft</td>
<td>$20,000</td>
</tr>
<tr>
<td>Total Income</td>
<td>$250,000</td>
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</tbody>
</table>

<table>
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<th>Fixed Costs</th>
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</tr>
</thead>
<tbody>
<tr>
<td>Admin &amp; General</td>
<td>$2,000</td>
</tr>
<tr>
<td>Heat, Lights &amp; Water</td>
<td>$14,000</td>
</tr>
<tr>
<td>Repairs &amp; Maintenance</td>
<td>$10,000</td>
</tr>
<tr>
<td>Staff Costs - 27 permanent local employees</td>
<td>$84,000</td>
</tr>
<tr>
<td>Vehicle Expenses – Fuel</td>
<td>$6,000</td>
</tr>
<tr>
<td>Vehicle Expenses – Repair &amp; Maintenance</td>
<td>$6,000</td>
</tr>
<tr>
<td>Sub-total</td>
<td>$122,000</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Variable Costs</th>
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</tr>
</thead>
<tbody>
<tr>
<td>Cost of marketing (Hunt and Tourism)</td>
<td>$5,000</td>
</tr>
<tr>
<td>Cost of hunts (Safari operator)</td>
<td>$60,000</td>
</tr>
<tr>
<td>Sub-Total</td>
<td>$65,000</td>
</tr>
<tr>
<td>Total Expenditure (fixed costs + variable costs)</td>
<td>$187,000</td>
</tr>
<tr>
<td>Profit</td>
<td>$63,000</td>
</tr>
</tbody>
</table>
DISCUSSION

Previous CCWR Initiative in Zimbabwe

This CCWR initiative follows an initial attempt to involve local communities in game ranching in the Mid Zambezi Valley (CIRAD 2000). Chivaraidze Game Ranch (CGR) was the first of its kind on communal land because the CAMPFIRE model was based on free-ranging wildlife (Chinhoyi 2004). The CGR covers an area of 3,200 ha in Ward 4 of Mbire RDC and it was fenced with the financial support of French government (http://www.ffem.fr). The ranch’s wild animals initially consisted of small populations of game; a restocking program was conducted from 1999 with the reintroduction of 500 impala (Aepyceros melampus) and a mixed population of 200 head of plains game (i.e., zebra [Equus quagga], wildebeest [Connochaetes taurinus], sable antelope [Hippotragus niger], tsessebe [Damaliscus lunatus], waterbuck [Kobus ellipsiprymnus] and eland [Taurotragus oryx]). The total investment was $350,000 USD mainly for restocking (33%), game fencing (21%) and building a butchery and offices (21%). With operating costs of $380 USD/km², the business model was based on the annual sale of 6.5 tons of fresh meat to the local community and of a few trophy animals allocated by ZPWMA, which represented two thirds of the game ranch revenues (LeBel et al. 2004). The institutional framework of the ranch was established in phases: in October 2000, the Chivaraidze Game Ranch became a CAMPFIRE project and transformed itself
in November 2003 into a cooperative company with an elected 12-member board of directors and 1,300 households of the ward as potential shareholders. With the interruption in 2005 of the shares allocation by the local elite, the cooperative company experienced internal instability. Once controlled by the ward leader and his cronies, most of the trained workers left the ranch, the performance of the ranch declined with the increase in poaching (Mombeshora and LeBel 200). A key lesson learned from this first CCWR initiative clearly indicates the main challenge to be more in the socio-political arena than in the technical capacity of the community to co-manage a commercial entity. The proposed M1 plus M2 partnership arrangement of the Mahenye project is an interesting set-up which will need to be translated into a robust and effective community-based institution (Chinhoyi 2004).

Implementing CAMPFIRE Principles
The Mahenye CCWR project highlights the fact that game ranching is a potential vehicle to foster CAMPFIRE principles (Taylor 2009b) and it reconciles wildlife management with equitable benefit sharing (Balint and Mashinya 2008).

Local governance.—With the partnership model, local ownership of wildlife will be strengthened. The community will have the power to control safari hunting in the CCWR and thereby efficiently utilize the hunting quotas issued by ZPWMA. The community will derive more revenue because they can ask safari operators to bid on a hunt by hunt basis.

Wildlife based revenues and benefits sharing.—Through direct payments from safari operators to local community, Mahenye will receive 96% of profit instead of the current 55% and it will increase gross revenues from a more diversified set of activities such as sport hunting and ecotourism. This will help to reduce conflict between the community and the Chipinge RDC.

Biodiversity conservation.—Fencing will reduce the occurrence of human-wildlife conflict, improving the social tolerance of local communities toward wildlife
It will foster the conservation of wildlife species in the Great Limpopo Transfrontier Park, which currently covers 35,000 km² (http://www.peaceparks.org).

**Fostering Community Buy-in**

In the case study of the Mahenye, the social challenge is great because the last decade ended up with the loss of honest brokers, leading the elite to capture the local CAMPFIRE project, the decline of revenues and ongoing conflicts between the RDC, the safari operator and the community (Rihoy et al. 2010). Fostering community buy-in in such a venture is not only a moral obligation but is critical for the success of the CCWR.

*Fencing and ownership.*—Fencing could be used as a practical tool to build the community’s ownership of the CCWR. With a 20 km game fence, each of the 960 households will be maintaining about 20 meters of game fence. The regular maintenance of the fence (clearing & repair) will be a practical and simple way to trade access to the CCWR products while monitoring the satisfaction level of the beneficiaries.

*Addressing basic needs.*—The challenge of such a wildlife enterprise is to address local community’s needs which are oriented more towards farming than wildlife issues (Cunliffe 2010). The Community Markets for Conservation model developed in Zambia provides a revolutionary approach that uses rural-based markets to support conservation (Lewis et al. 2011). Through a trading mechanism, farmers neighboring the game ranch will be contracted by the CCWR to use climate-smart agricultural practices. In return they will be rewarded with better post-harvest handling and trading conditions, including better prices; the CCWR acting as an agri-commodity broker.

**A Local Hub Attracting Funders**

Promoting CCWR in communal lands raises the challenge of attracting funding in a capital expenditure exercise of $5,000-10,000 USD/km² in an unsecure economic environment (Lindsey et al. 2011). Nevertheless, examples from commercial wildlife ranching in South Africa should guide this process as it’s clearly demonstrating that this land use option offers a better economic output in marginal land up to $ 2,500 USD/km² (Dry 2011).

**The Carbon Credit Dream**

Earning carbon credits through avoided deforestation could be particularly relevant for Zimbabwe which has a high deforestation rate of about 2% per year (http://rainforests.mongabay.com/deforestation/2000/Zimbabwe.htm). The Miombo woodland soils in CCWR are capable of storing more than 100t C per ha (Williams et al. 2008). If a community carbon project in Mozambique enables a $35 USD cash payment per ha over seven years for carbon sequestered (Jindal et al. 2008), its impact on a 100 km² CCWR willing to avoid a deforestation rate of 2% will be limited to $1,000 USD per year, a fraction of its income.

**MANAGEMENT IMPLICATIONS**

The most challenging issue is the functionality of the partnership arrangement. Such levels of revenue generate a lot of expectations at household’s level and are often perceived by the local elite as handy tools to reinforce their power. A careful mediation process by a third party, an honest broker (Taylor 2009b), will be needed to reach a fair and balanced deal between all partners. This process will take time and needs training not only of the classic beneficiaries but also for decision makers in the public and private sectors.

**ACKNOWLEDGMENTS**

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RESTORATION AND WILDLIFE CONSERVATION AS AN ECONOMIC INCOME ALTERNATIVE

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ABSTRACT: Conservation impacts of human-wildlife conflicts indicate an urgent need for effective solutions that will reduce the killing of wildlife. Protected areas are just a partial solution, and effective conservation demands some means of guaranteeing the coexistence of people and wildlife outside protected areas. Nature-based tourism is a management alternative that can provide an income for landowners and increase incentives for wildlife conservation. This work focuses on a case study in degraded lands surrounded by primary forest in the Selva Lacandona, in southeast Mexico. In this region, we are helping local people assess the viability of a project that links restoration and wildlife monitoring programs with tourism and wildlife viewing. First, we implemented a restoration program in those places with land degradation and inventoried different large- and medium-sized mammals that inhabit the area. Finally we developed an ecotourism proposal as an alternative for preserving natural and cultural resources.

KEY WORDS: ecotourism, Lacandona, mammal monitoring, Mexico, perforation, rainforest restoration, land use change, wildlife.


Nearly 60% of all of the species on Earth inhabit tropical rainforests, making it the most diverse biome on the planet (Hill and Hill 2001). In Mexico, the biggest remnant of rainforest is located in the region known as Selva Lacandona in Chiapas, that together with Calakmul in Campeche, and Peten in Guatemala and Belize represents the largest area of tropical forest in Central America (Medellin 1994).

The Lacandona rainforest originally covered over 1.8 million ha, but due to human activity, in <40 years it has been reduced to ca. 500,000 ha, including the 331,200 ha covered by the Montes Azules natural protected area (INE 2000). There are several communities near this biosphere reserve that are intensively transforming the rainforest into grazing and agricultural lands. Some of these communities belong to the municipality of Marques de Comillas, where this study was conducted.

Despite the intense deforestation, there are still some large (<10,000 ha), well-preserved fragments of rainforest outside the natural protected areas inhabited by several endangered wildlife species, such as jaguar (Panthera onca) and tapir (Tapirus bairdii; Bolaños and Naranjo 2001). These fragments are also ecologically important because they provide connectivity between reserves and are therefore in need of conservation (INE 2000). However, because these rainforest fragments are owned by local communities, they are at risk of being converted to other land uses. One of the main transformation processes occurring on these lands is the emergence of perforations, which occurs when anthropogenic holes or gaps are made in the interior of intact forest fragments; this kind of deforestation causes potential edge effects deeper in the wildlife communities (Ritters and Coulston 2005).

Specifically in the region of Marques de Comillas, 50% of deforestation is originated by perforations. The outcome is a fragmented landscape where cattle and agricultural ranches are established, creating potential areas of human-wildlife conflict. Due to management difficulties and low production rates, most of these pasturelands are abandoned resulting in patches of transformed vegetation surrounded by primary forest (Carabias et al. 2009).

Ecotourism could be an alternative, providing an economic incentive for restoration and conservation of such abandoned lands (World Travel and Tourism En-
Theoretically, ecotourism offers tangible economic benefits from wildlife as it offsets the costs of protection and coexistence, providing enough profits to the local community so that the inhabitants will value and protect their wildlife heritage (Goodwin 1996). Given the urgent need for generating mechanisms for preserving ecosystems outside natural protected areas, where communities feel engaged with the whole conservation process, we are helping local owners in the development of an ecotourism project in an abandoned perforation. Our main objective is to combine a restoration program and a community wildlife monitoring program with ecotourism activities as an alternative economic opportunity for local landowners thus increasing the value of wildlife. To accomplish this, we are focusing our efforts on the restoration of a perforation that is surrounded by a preserved rainforest fragment inhabited by several charismatic mammal species. Simultaneously, we are monitoring medium and large mammal populations using camera traps in the areas adjacent to the deforested patch. Our long-term goal is to link both projects with ecotourism by promoting wildlife viewing visits, so that local wildlife will serve as the main economic income of local inhabitants.

STUDY AREA
This study was conducted in a 37 ha perforation located at the north of Boca de Chajul, Marques de Comillas, Mexico (Figure 1). Mean annual precipitation was approximately 3,000 mm, while mean annual temperature was 25 °C with differences between summer and winter temperatures of <5 °C (Siebe et al. 1991). The deforested area was located inside a 750 ha fragment of intact primary forest. Primary forest vegetation was typical of the Lacandon rainforest including tree species such as guasiban (Albizia leucocalyx), guapaque (Dialium guianense), ramon (Brosimum alicastrum), plumillo (Schizolobium parahybum), ceiba (Ceiba pentandra), and cedro (Cedrela odorata). In the understory, palms such as chocho (Bactris baculifera) and tree ferns (Cyathea spp.) are widespread (Carabias et al. 2010). The perforation was cleared of natural vegetation and several exotic grass species (e.g., Brachiaria decumbens and Echinochloa polystachya) were introduced 14 years ago. It was used as pastureland for 12 years, until the paddock was abandoned 2 years ago because of livestock predation by jaguars. After 2 years of abandonment, the site was covered mostly by exotic grasses and secondary vegetation with species such as Bellucia grossularoides, Eugenia spp., Xylopia frutescens and Casearia spp.

METHODS

Restoration
To alleviate the degraded environmental conditions and accelerate the ecological regeneration of the site, a restoration program consisting of reforestation of the 37 ha with native tree species will be conducted. First, in order to assess the potential for meeting our restoration objectives, we conducted a feasibility study from September 2011 to July 2012. We defined 2 experimental units inside the perforation: A and B. Both sites have deficient drainage and soil aeration problems. These unfavorable conditions for plant growth are accentuated in unit B, which is periodically flooded. We planted native trees species in a density of 650 trees/ha as recommended by the National Forestry Commission, with species selected for each area based on their tolerance to flooding. To increase plant survival, we removed exotic grasses manually. We then measured the survival rate by counting the total number of living individuals each month for plumillo, ramon, frijolillo (Cocoba arborea), hule in unit A and amate (Ficus cotinifolia), guatope (Inga vera), guasiban and hule in unit B.

Wildlife Monitoring
Camera traps were used to identify mammal species > 1kg present in the restoration area and the surrounding primary forest (Medellin 1994). Five camera traps were placed in different locations from April to July 2012: 1 in a forested area bordering the “Negro”
stream that divides the perforation forming a vegetated corridor (camera 1; Figure 1) where tapir tracks were constantly seen, 1 in the experimental restoration unit B (camera 2; Figure 1) and the rest inside the primary forest surrounding the restoration site (cameras 3, 4 and 5; Figure 1). Each camera operated during 3 sampling periods of 20 days, resulting in a total sampling effort of 300 trap-days. To determine if camera trapping could be simultaneously used as a tool for wildlife monitoring and ecotourism activities, we calculated the probability of capturing any of the observed species during the first 5 and 10 days of each survey.

**Design and Cost Analysis of the Ecotourism Project**

We developed a proposal for an ecotourism project where the principal attraction is the participation in wildlife monitoring activities, the use of camera traps and tracking, and wildlife viewing. As a first step to assess the viability of this ecotourism project, we calculated the initial investment cost, which includes the infrastructure value, purchase of equipment, and local landowner training. We then estimated the annual operational cost, which includes maintenance of infrastructure, purchase of equipment and supplies, staff salaries, and publicity costs. This exercise allowed us to estimate anticipated profits with the goal of obtaining a minimum benefit/cost ratio of 1.3 or 30% profit. All the prices were based on market prices from Comitan de Dominguez and include the cost of transportation to Boca de Chajul. Initially, the estimations were calculated in Mexican pesos (MX) and then converted to American dollars (USD) at a rate of $1 USD = $13.23 MX (Banco de México).

**RESULTS**

**Restoration and Wildlife Monitoring**

Nine months after planting, saplings had a global survival rate of 74% in unit A and 72.5% in unit B. In unit A, the most successful species were *S. parahybun* and *C. arboarea*, with a survival of 85% and 90% respectively, while *B. alicastrum* had a survival of 65%. In unit B, *I. vera*, *F. cotinifolia* and *A. leucocalix*, had a high survival (75, 80 and 85%, respectively). However, *C. elastica*, in both units, had a survival rate of about 50%.

The trapping effort of 300 trap-days resulted in photographic captures of 10 different species, with 6 species photographed more than once (Table 1). None of the species were photographed both in pasture land and rainforest. The only species photographed in pasture land was white-tailed deer (*Odocoileus virginianus*) which was also the species most frequently photographed. The rest of the species were only photographed inside of the primary vegetation fragment which was farthest inside the forest. Camera 1 had no photographs, but this could have been due to camera malfunction because tapir tracks were found in the area.

In each sampling period, a picture was taken during the first 5 days of survey; 4 species, including the jaguar, were photographed during this first period. After 10 days, 80% of the 10 species were photographed. This result could imply that using camera traps as part of the ecotourism attraction is viable in terms of species photographic probabilities during periods when tourists would be visiting the site.

**Design and Cost Analysis of the Ecotourism Project**

We proposed the construction of a visitor center near the restoration site for tourists. Guides would take tourists on jungle walks from the visitor center. Tourists will be taught the basics of monitoring techniques, camera trap placement and conduct intensive searches for animal tracks. Construction of a blind will allow tourists to view wildlife and enjoy the ecosystem. After at least 5 days, tourists can collect memory cards from the camera traps and take their photographs with them. It’s important to mention that this project was designed with the goal of integrating it with other strategies of local development, as complement of tourism activities such as places for lodging.

**DISCUSSION**

An ecological restoration program assists the recovery of an ecosystem that has been degraded, damaged or destroyed (SER 2004). This is a long-term process and its implementation cost is high. Conservation programs for wildlife in this region need to account for the fact...
that most of the land outside the natural protected areas is owned by local communities who are transforming the ecosystem according to their economic and cultural needs (Cardona 2008). The success of this restoration process depends on a multidisciplinary approach that considers social, economic and environmental needs. Profits from the restoration program must directly benefit local communities in order to assure its continuity. Otherwise, people may lose interest and abandon the program. To establish this project the Mexican government is currently providing start-up financing through 2016. Therefore, it is important that we develop the project to integrate the success of restoration activities and the conservation of wildlife while simultaneously enabling local communities to participate in planning, decision-making, and the implementation and development of economic alternatives to land use change.

Community-based ecotourism can be regarded as a tool for sustainable local development in rural areas (Himberg 2004) and might be a partial solution for the problem of rainforest conversion in this region, especially considering the presence of charismatic wildlife species that were documented in the rainforest fragments near the restoration site. Wildlife ecotourism and monitoring using camera traps, provides a unique and innovative approach for tourists to participate in wildlife monitoring and to detect secretive animals during their visits. Furthermore, based on our results, there should be a high probability of photographing an animal during the first 5 days of the survey and this probability will increase if more cameras are deployed so that tourists might be able to capture photographs of these animals in a relatively short trip to the area.

In any case, if tourism is to act as an economic incentive for tolerance of wildlife, then it must generate profits that offset the direct and indirect cost of living with wildlife (Walpole and Thouless 2005). Nevertheless, our calculations for the initial investment cannot be financed by local landowners due to their economic conditions, therefore external funding is required, which may slow the development of the project.

Michaelidou et al. (2002) suggested that “ecosystem and community viability are interdependent, so efforts to enhance one dimension would be unsuccessful if the other dimension is ignored”. In this manner we are simultaneously encouraging the restoration process and protection of wildlife by generating income for the landowners from tourism activities. But in order for this to actually succeed, economic profit should be higher than operational costs (Goodwin 1996). Nonetheless, the results of this preliminary study are encouraging, but we currently need to develop a more comprehensive development plan and economic analysis to ensure the success of the project. Along these lines, more work with the local communities must be done.

**MANAGEMENT IMPLICATIONS**

As human development expands, people and wildlife are in greater direct competition for a shrinking resource base, and human-wildlife conflict is becoming more widespread and persistent. The development of a project that incorporates wildlife study with camera traps and tracking as a tourism attraction could be a way of mitigating this conflict. This project will demand that landowners be trained in monitoring techniques and wildlife management. The final goal is to increase the value of native wildlife by creating a sustainable project where visitors will support the community not only with their payments for tourism-related expenses, but also with the collection of scientific data that will help in understanding long-term population dynamics and the community responses to vegetation restoration projects.

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**LITERATURE CITED**


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PREDATION MANAGEMENT IN THE UNITED STATES: THE FEDERAL WILDLIFE SERVICES PROGRAM

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ABSTRACT: Predation management is a necessary but sometimes controversial activity. When conducted for the protection of livestock and native wildlife, predation management includes the use of both lethal and nonlethal methods by landowners and government agents. The role of the government is as much to provide oversight as it is to save livestock and protect wildlife. Within the U.S., predation management is conducted at the federal and state level by the U.S. Department of Agriculture-Animal and Plant Health Inspection Service-Wildlife Services program. The program provides federal leadership and policy while incorporating state and local funding to balance the needs of agriculture with the environment. This paper discusses the Wildlife Services program for predation management with emphasis on methods, funding and cooperative relationships including the protection of native wildlife and the economics of predation management.

KEY WORDS coyote, predation, predators, Texas, wildlife damage management, Wildlife Services.


In the United States, wildlife is not owned by individuals, but wildlife is held by the government in public trust for all (Organ et al. 2010). Certain species, such as migratory birds, marine mammals and threatened or endangered species are managed under additional federal, state and/or tribal authorities due to legislative direction. Other, generally resident, wildlife species are managed by state or tribal wildlife agencies. Wildlife is considered a highly valuable public resource (U.S. Fish and Wildlife Service and U.S. Census Bureau 2006) and wildlife management programs generally seek to maximize the presence of native wildlife. However, wildlife at any population level can cause conflicts with public and private interests (Conover et al. 1995). Restrictions are in place which prevent or restrict private landowners from controlling wildlife damage in a manner which will negatively affect the environment (e.g., Bald and Golden Eagle Protection Act, Endangered Species Act, Federal Insecticide Fungicide Rodenticide Act, Migratory Bird Treaty Act). Just as we don’t allow a private timber company to manage the National Forests, we don’t allow livestock owners, for example, to manage predator populations through unregulated means. The role of the government in this instance is to protect the environment while assuring the appropriate protection of private property from public wildlife damage.

The U.S. Congress recognized the need for a federal wildlife damage management program to mitigate human wildlife conflicts due to publicly owned wildlife. In 1885, Congress established our parent program, the Section of Economic Ornithology within the U.S. Department of Agriculture, under Dr. C. Hart Merriam (Hawthorne 2004). The program is currently administered within the U.S. Department of Agriculture, Animal and Plant Health Inspection Service (USDA-APHIS) under the title of Wildlife Services (WS). Wildlife Services operates both an operational wildlife damage management program and a program for research into wildlife damage conflicts and impacts as well as the ecology of wildlife species causing damage. The WS National Wildlife Research Center, headquartered in Fort Collins, Colorado, USA is the only facility wholly dedicated to research on human-wildlife conflicts.

The operational portion of the WS program includes federal employees in every state and territory to respond to problems caused by wildlife. Congress recognized that the state governments may also operate wildlife damage programs and provided the authority for WS to cooperate with each state in the resolution of such conflicts (7 USC 426-426c). Because the various state and tribal management authorities also have the responsibility for the resolution of conflicts, the exact
structure of the WS program varies between the different states, territories and tribes. In some states, the federal wildlife damage management program may be limited to surveillance and control of wildlife borne diseases of national or regional concern. In other states, the state wildlife agency may contribute funding to support a cooperative program wherein state and federal trappers work side-by-side to address conflicts caused by specific species of wildlife. In many states, the state agency will address conflicts caused by protected species such as game animals while the federal program will address conflicts caused by unregulated, primarily predatory wildlife (e.g., coyotes [Canis latrans]). The WS program also has a responsibility to manage damage caused by zoonotic diseases, migratory birds, feral animals, invasive species, and other vertebrates and by certain populations of reintroduced, endangered predators (i.e., wolves [Canis lupus]).

THE WILDLIFE SERVICES PROGRAM

Predation management was historically the largest single function of the WS program (Hawthorne 2004). Large predators, such as gray wolves (Breck and Meier 2004) and grizzly bears (Ursus arctos; Brown 1996) were almost completely removed from western states largely due to government programs prior to the 1960s. The growing environmental movement in the 1960s and early 1970s led to reforms in the program (Leopold et al. 1964, Cain 1972, Feldman 2007). Today, the focus of predation management is to reduce predation impacts on private and public resources to acceptable levels without negatively impacting the populations of native predators. The main predators addressed by the predation management program include coyote, bobcat (Lynx rufus), mountain lion (Puma concolor), and black bear (Ursus americanus). As populations increase and conflicts occur, WS is often charged with management of predation by endangered predators. Among these, WS conducts locally important programs on gray wolves and grizzly bears. Programs also exist in some states for the management of mesocarnivores, including raccoon (Procyon lotor), gray fox (Urocyon cinereoargenteus), red fox (Vulpes vulpes), Arctic fox (V. lagopus) and feral predators including feral dog (Canis lupus familiaris) and feral cat (Felis catus).

Predation management is applied for the protection of livestock, natural resources, such as vulnerable populations of native wildlife, or for the protection of human health and safety. Some species, such as coyotes, may be removed for livestock protection, but also serve as valuable sentinel species for diseases such a bubonic plague, bovine tuberculosis or rabies.

INTEGRATED WILDLIFE DAMAGE MANAGEMENT

The WS program uses an integrated wildlife damage management (IWDM) approach to predation management. Like Integrated Pest Management (IPM), IWDM seeks to integrate nonlethal, lethal, education, and research methods into resolving wildlife damage situations. In some cases, such as wolf, mountain lion or black bear damage management, the focus of the project is to target a single, offending individual or local group of individuals only after losses have occurred and been verified by governmental officials. Other species, such as coyotes or raccoons, may be removed prior to the onset of losses given that losses have occurred in the past and are reasonably expected to occur again. This approach recognizes that some species are more vulnerable to mortality than others and seeks to minimize impacts to low density predator populations. In some cases where losses must occur before control efforts are authorized, compensation from government or private funds is available to compensate livestock producers who lose livestock to these predators (Wagner et al. 1997).

Integrated wildlife damage management differs from traditional IPM in that the native predators removed are considered pests by affected landowners, but they also serve a role in the ecosystem. Integrated wildlife damage management seeks to balance the potential negative ecosystem impacts of predator management with the positive economic benefits of reducing predation. In many cases, predation impacts can be both negative and positive. Two examples may serve to illustrate the balancing act WS must consider when addressing predation management:

In western Utah, domestic sheep (Ovis aries) graze on open rangeland accompanied by sheep herders in the winter. These sheep generally stay in the valleys between mountain ranges because water is a limiting factor and it must be trucked to the sheep in the absence of snow. While sheep are in the valleys, coyotes are the main predator and WS conducts both preventive and corrective coyote control. However, when it snows, the sheep herders will often take sheep into the mountains to take advantage of snow and otherwise unavailable forage. Mountain lions exist in low densities in these mountains and prey on overabundant herds of feral horses (Equus caballus). If a mountain lion were to kill a domestic sheep, WS has the authority to remove the lion to prevent further losses. However, killing an individual lion in these cases may exacerbate feral horse management, allowing horse populations to increase further. In these cases, WS may work with the sheep herder to move the sheep from the predation...
area to prevent additional losses without necessitating mountain lion removal.

In other cases, WS may be tasked with protecting livestock in an area with depressed populations of native wildlife. While the standard approach to livestock protection includes both lethal and non-lethal methods, some non-lethal methods, such as guard dogs, may exacerbate predation on native wildlife by concentrating predators and native wildlife in smaller habitats. In these cases, WS may choose to implement a geographic area designed to protect both the wildlife and livestock. Close coordination with other wildlife agencies provides WS with the information necessary to coordinate predation management activities with multiple resource agencies.

The WS program also addresses predation conflicts with native wildlife. High priorities are given to protecting endangered wildlife such as black-footed ferret (Mustela nigripes), Columbian white-tailed deer (Odocoileus virginianus leucurus) and many colonial nesting bird species. Predation management is applied for game management when species are considerably below management objectives and predation appears to be holding these populations in a “predator pit.”

WILDLIFE DAMAGE MANAGEMENT METHODS USED

Wildlife Services uses a variety of methods for the protection of vulnerable resources. Methods may be implemented prior to the initiation of losses (i.e., preventive control) or after losses have occurred (i.e., corrective control). In general, preventive control is implemented for non-protected predator species with high populations and reproductive rates, such as coyote, while corrective control is implemented for those species with limited populations or where the species is more vulnerable to human caused mortality, such as wolves and bears. Commonly used non-lethal methods include the use of chemical capture drugs, hazing or frightening devices, increased vigilance/activity during vulnerable periods, live-trapping, temporary fencing or fladry for small confined areas and training for livestock guardians. Livestock producers are also encouraged to implement non-lethal husbandry methods including changing the timing of livestock births, avoiding grazing areas where losses are most likely, using guard animals or herders, night penning, confined birthing, predator resistant fencing and various habitat management practices. In many cases, accelerated non-lethal methods may not be economically feasible.

Methods implemented by Wildlife Services may be either lethal or non-lethal, depending on the management objectives for the animal targeted. Wildlife Services uses foothold traps, foot snares, neck snares, and live traps to capture depredating wildlife. Some species (e.g., black bears, mountain lions and bobcats) may be trailed with trained dogs and bayed in trees or on the ground. In some cases, animals are captured and relocated, in others the animal is euthanized following capture.

Some predation management methods are inherently lethal in nature. Toxicants, including sodium cyanide in the M-44 device and sodium monofluoroacetate (Compound 1080) in the livestock protection collar, are registered for coyote control. Wildlife Services has also registered a den fumigant, which produces carbon monoxide when burned, to remove predators in dens. Other lethal methods include shooting (including calling and shooting) and aerial shooting from fixed wing and rotary wing aircraft.

Predator research by WS is headquartered at our National Wildlife Research Center’s Field Station in Logan, Utah. Researchers have the ability to test tools with captive coyotes in pen facilities prior to testing applications in the fields. Additional research is conducted by researchers at the Center’s headquarters in Fort Collins, Colorado, or on location throughout the U.S. Wildlife Services works with non-governmental organizations, schools, extension, universities, and one on one to provide knowledge and techniques for producers to help themselves or to educate the public about predator-prey relationships.

RESOURCES PROTECTED AND LOSS DATA

Understanding predation loss data is important to decision makers. Most surveys conducted by researchers indicate losses with predation management programs in place (Knowlton et al. 1999). Livestock loss data in the absence of predation management has been studied in a few areas (Knowlton et al. 1999, Shwiff and Bodenchuk 2004). Nationally, 1.73% of adult sheep and 3.86% of all lambs were lost to predators in 2004 (USDA 2007). However, these losses represent sheep operations in areas with predation management in place. Bodenchuk et al. (2002) reported on 5 studies that examined losses of sheep and lambs to predators in the absence of predation management. Losses of adult sheep to predators ranged from 1.4% to 8.1% and averaged 5.7%. Lamb losses, in the absence of predation management, ranged from 6.3% to 29.3% and averaged 17.5%. Similar predation losses have also been reported for Wyoming (Taylor et al. 2009). It is important to note that these studies were designed to monitor sheep losses primarily to coyotes in the western U.S. following the removal of large scale toxicant use in 1972. The data probably underrepresent sheep and...
lamb losses in the absence of predation management for areas with multiple predator species in addition to coyotes.

Bodenchuk et al. (2002) also reported on 2 studies which examined predation on goats in the absence of predation management. In a 2-year study in Texas, adult goat losses averaged 49% and kid goat losses averaged 64% (range 33-95%). In a second, short-term study, 00% of all kid goats were killed when fencing alone was used for predation management. Bodenchuk et al. (2002) used 50% goat losses as a conservative loss figure to represent predation on range goat operations in the absence of predation management.

Beef cattle losses to predators have been examined in a number of studies. Overall, predation on adult beef cattle is low, estimated at 0.23% (USDA 2012). Only 4.2% of adult beef cattle losses are predation related. Beef calf predation losses are estimated at 1.44% of the beef calf crop. Nationally, 4.2% of the adult beef cattle losses (to all causes) and 15% of all calf losses were due to predation. Predation is not evenly spread among producers and the majority of cattle producers experience little to no predation-related losses. However, for those producers who do experience predation losses, they can be significant. In Utah, for example, beef cattle producers who experienced losses averaged 3.6% of their calf crop in the absence of predation management. In another study site in Arizona (with predation management in place), predation accounted for 5.5% of the accountable calf crop (Breck et al. 2011).

Wildlife Services significantly reduces predation for those ranchers with cooperative agreements in place. Predation losses with management in place averaged 1.6% of adult sheep, 6% of lambs, 12% of goats and less than 0.5% of the calf crop. Bodenchuk et al. (2002) calculated direct benefits (using 1999 losses and prices) at $62.6M.

MANAGEMENT OF PREDATORS

While a number of predators affect livestock, nationally coyotes are the greatest single predation threat. Coyotes are responsible for 56.9%, 64.2% and 51.7% of calf, lamb, and adult sheep predation losses, respectively (USDA 2007, 2012). Regional differences occur, with wolf losses occurring in the Great Lakes, Southwest and Rocky Mountain regions, while bear and mountain lion losses are largely confined to the western half of the U.S. Whereas losses to larger predators may not be significant on the national scale, they can be locally or regionally important. Breck et al. (2011) noted that 79.5% of the calf predation losses in one study site were due to mountain lions even though other predators were in the area. In a mixed predator complex, the USDA (1996) noted that 40% of the lamb losses in Utah were due to mountain lions and bears combined.

While the WS program implements an IWDM program (Table 1), many of the available non-lethal methods are primarily implemented by livestock producers. Wildlife Services expertise is necessary to effectively address predation with the least impacts to predator populations while remaining effective for the protection of livestock, vulnerable wildlife and human health and safety.

PROGRAM ORGANIZATION

When establishing the federal WS program, the U.S. Congress noted that individual states also had an interest in predation management and encouraged the federal program to “cooperate with States, local jurisdictions, individuals, public and private agencies, organizations and institutions” (7 USC 426-426c). Such cooperation varies across the U.S., depending largely on the level of state and private funding dedicated to similar purposes and the vulnerability of livestock. In some states, WS employs federal wildlife technicians, often locally called “trappers”, to implement the program. In others, notably Utah, Nevada, New Mexico, Oklahoma and Texas, both federal and state personnel are employed, with supervision, directives and significant resources provided by the federal WS program. While funding and cooperative relationships vary across the different states, the Texas Cooperative WS program provides an example.

The Texas Example

In Texas, WS provides funding and leadership for a cooperative wildlife damage management program. In addition to predation management, wildlife damage

<table>
<thead>
<tr>
<th>Species</th>
<th>No. states</th>
<th>Lethal removal</th>
<th>Nonlethal removal</th>
<th>Number dispersed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Black bear</td>
<td>18</td>
<td>574</td>
<td>599</td>
<td>42</td>
</tr>
<tr>
<td>Grizzly bear</td>
<td>1</td>
<td>4</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Feral cat</td>
<td>37</td>
<td>1277</td>
<td>806</td>
<td>201</td>
</tr>
<tr>
<td>Coyote</td>
<td>44</td>
<td>83242</td>
<td>65</td>
<td>515</td>
</tr>
<tr>
<td>Feral dog</td>
<td>31</td>
<td>465</td>
<td>199</td>
<td>97</td>
</tr>
<tr>
<td>Arctic fox</td>
<td>1</td>
<td>102</td>
<td>27</td>
<td></td>
</tr>
<tr>
<td>Grey fox</td>
<td>18</td>
<td>2539</td>
<td>110</td>
<td>4</td>
</tr>
<tr>
<td>Red fox</td>
<td>39</td>
<td>1897</td>
<td>44</td>
<td>336</td>
</tr>
<tr>
<td>Mountain lion</td>
<td>11</td>
<td>402</td>
<td>19</td>
<td>3</td>
</tr>
<tr>
<td>Bobcat</td>
<td>19</td>
<td>1263</td>
<td>57</td>
<td>7</td>
</tr>
<tr>
<td>Wolves (all)</td>
<td>7</td>
<td>365</td>
<td>52</td>
<td>0</td>
</tr>
</tbody>
</table>
addressed includes feral swine (Sus scrofa domesticus) damage to agriculture and crops, beaver (Castor canadensis) damage to waterways and agriculture, several different issues related to migratory birds such as bird-aircraft strikes and surveillance and management of zoonotic diseases, including rabies. The federal program provides overall direction and supervision, significant funding and 4 aircraft (2 fixed wing and 2 helicopters) for wildlife damage management. Wildlife Services maintains pesticide registrations with the Environmental Protection Agency and Texas Department of Agriculture for toxicants used for predation management.

The State of Texas also contributes significantly to the overall cooperative program. The state, through the Texas A&M University System and AgriLife Extension Service, contributes funding and administrative support for the cooperative program. Most of the field technicians are state employees. Other state agencies, such as the Texas Department of Agriculture and Texas Parks and Wildlife Department provide financial assistance, regulatory oversight for pesticides and support for cooperative projects involving predation management.

Individual rancher and county level support exists where predators impact livestock. A statewide, non-profit association, the Texas Wildlife Damage Management Association (TWMDA), collects cooperative funding into a Wildlife Damage Management Fund, which is used by the cooperative program to fund aspects of the predation management program.

In practice, the program operates efficiently, using a combination of federal, state, county, and private funds to provide services. A 3-party Memorandum of Understanding establishes the cooperative relationship and provides the overall framework for cooperative programs. Individual technicians are hired (largely as state employees) at the county level to conduct predation management activities for cooperating ranchers. The program does not have regulatory authority to enter property, so cooperating ranchers must provide access to property and concur with species taken and methods to be used.

The Texas program costs an average of $5,400 U.S. per month to keep a full-time employee in the field. The TWMDA collects $2,400 per month per employee from cooperating counties and/or ranchers to cost-share these employees. The TWMDA pays certain expenses to both the USDA and the Texas AgriLife Extension Service. The remaining costs, about $3,000 per month, are paid from state and federal appropriations (Figure 1).

Policies utilized by the cooperative program are developed by USDA and implementation is overseen by a USDA employee (i.e., the State Director). To facilitate administrative functions at the state level, the USDA State Director also serves the State of Texas as a program leader (an ex officio position) for AgriLife Extension Service. The state is divided into 9 Districts, each with a field employee as a District Supervisor (Figure 2). Seven of the 9 District Supervisors are federal employees, while 2 are state employees. Because the aircraft are federal assets, the aircraft pilots are federal employees. Compensation systems for state and federal employees are negotiated between the 2 parties to assure that employees are paid equally for equal work. In Fiscal Year 2012, beginning
technicians (trappers) were paid approximately $24,000 U.S. annually.

Technicians are empowered to address wildlife damage management within their respective counties or assignments. In much of the state, technicians deal predominantly with livestock predation, with special emphasis on sheep and goat protection from coyotes and bobcats. In western Texas, wildlife protection from predators is more of a concern, so while methods are similar, the degree of control necessary varies. Feral swine also pose a risk in Texas and as a result, technicians often become involved in multiple species projects. In Fiscal Year 2011, the Texas WS program removed 961 bobcats, 20,464 coyotes and 24,680 feral hogs statewide (Figure 3).

MANAGEMENT IMPLICATIONS
While the cooperative approach provides logistical challenges, it also provides an exceptional balance between producer-led options for predation management and societal needs for wildlife protection. In Texas, livestock producers and counties pay approximately 25% of the total program cost. State and federal funding provides the balance, recognizing that wildlife can damage private resources, but also providing protection for wildlife, even predator species, through a publicly funded, accountable program. Utilizing parallel funding allows the WS program to choose the appropriate employment status for employees, provides multiple purchasing options for necessary supplies and equipment, and allows resources to be deployed to where the need exists rather than being bound by administrative rules from a single agency. The fact that counties and producers continue to assess themselves for a portion of the cost for these services is proof of the value of the program to the economy of the rural U.S. and the individual wildlife resources the government protects.

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MANAGING HUMAN-WILDLIFE CONFLICTS ON THE “HARD EDGES” SYMPOSIUM AND PANEL DISCUSSION

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ABSTRACT: Boundaries of national parks, preserves, refuges, and privately-owned lands, often called “hard edges,” raise unique and significant challenges for managing human-wildlife conflicts because of the diverse and often opposing opinions of stakeholders with interests inside and outside of the boundaries. Politics, economic development, human health and safety, livestock depredation, wildlife use, and human dimensions influence management decisions and their effectiveness in resolving problems. We conducted a symposium and panel discussion on Managing Human-Wildlife Conflicts on the “Hard Edges” at the 4th International Wildlife Management Congress in 2012 to discuss issues and identify types and causes of human-wildlife conflicts, reveal associated policy and legal issues, promote potential solutions, and increase communication among interested stakeholders. Over 150 Congress attendees participated in the sessions that lasted nearly 3 hours. From the presentations, question and answer session, and summary notes, we generated 3 lists of topics that warrant further discussion. Associated overarching issues were identified, which include: the nature of hard edges; complexities of conflicts; key ecological, social-economic, and political factors that affect conservation and conflict; human land-use patterns and associated distributions of wildlife; human attitudes, tolerance levels, and behavior; site- and species-specific management strategies; and cost-effective management options. Unfortunately, human-wildlife conflicts at the hard edges seem to be increasing in frequency, scope, and importance, which influence wildlife management at local and global scales.

KEY WORDS: hard edges, human health, human-wildlife conflicts, large carnivores, livestock predation, megafauna, national parks, preserves, refuges, wildlife damage management


Management of wildlife resources along “hard edges” of national parks, preserves, refuges, and privately-owned lands is complex, arduous, and controversial, whether the boundaries are physical (e.g. fences), jurisdictional, or political (Newmark et al. 1993, Treves 2007, Andrew-Essien and Bisong 2009, Lamarque et al. 2009). This is especially true, given today’s diverse public opinions, special interests, and government responsibilities. Issues related to human well-being and conservation of wildlife resources including politics, human dimensions, health and safety, wildlife harvest, economics, “traditional use,” and available options for wildlife damage management clearly influence management decisions and their effectiveness in resolving problems. Disparity often exists in the distribution of positive and negative impacts among constituents on either side of hard edges (Newmark et al. 1993, Woodroffe et al. 2005, Lagendijk and Gusset 2008, Sifuna 2010). For example, in sub-Saharan Africa, safari operators, concessionaires, and government officials may reap economic benefits of ecotourism associated with wildlife, such as elephants (Loxodonta africana), lions (Panthera leo), and leopards (P. pardus) within national parks while local residents adjacent to these parks experience threats to human health and safety and economic losses to wildlife that move from parks onto private or communal lands (Kiiru 1995, Patterson 2004, Patterson et al. 2004, Bauer and De lough 2005, Kolowski and Holecamp 2006, Holmern et al. 2007). In a recent book titled, “Save Me from the Lion’s Mouth: Exploring Human-Wildlife Conflict in Africa,” James Clarke stated, “The challenge to conservationists is no longer simply a case of saving our wildlife heritage. By raising funds to put up fences and aiding zoological research they have done wonders – but little is being done to win the hearts and minds of those outside the reserves so they feel safer; so that they receive compensation for the loss of livestock, crops, and lives to wild animals; so that they perceive wildlife in a positive light and receive tangible benefits from its presence. The responsibility of wildlife professionals and their agencies, organizations, and institutions is to work with people with potentially divergent views to promote sustainable natural resources and if necessary, resolve conflicts between wildlife and human interests.
that lead to coexistence of humans and wildlife” (Clarke 2012, page 4). Populations of tigers (P. tigris) and elephants (Elephas maximus) residing in protected areas of southern and southeastern Asia often traverse beyond park and preserve boundaries, threatening lives and livelihoods of residents on adjacent lands (Gubbi 2007, Ogra and Badola 2008). In North America, recovering populations of gray wolves (Canis lupus) frequently are in conflict with humans because of both real and perceived predation on livestock, pets, and desired wildlife species, such as elk (Cervus canadensis; Bangs et al. 1998, Bradley and Pletscher 2005). In northern Europe, increased vehicle collisions with moose (Alces alces) are perceived a consequence of overabundant populations of moose in national parks and preserves (Groot Bruinderink and Hazebroek 2002, Seiler 2004).

The theme of the 4th International Wildlife Management Congress (IWMC) in Durban, South Africa in 2012 was “Cooperative Wildlife Management Across Borders: Learning in the Face of Change.” The issue of human-wildlife conflicts at the hard edges fits perfectly within this theme due to its complexity, significance, and global relevance. We wished to host a public forum on the subject to: (1) identify types and causes of human-wildlife conflicts, (2) reveal associated policy and legal issues, (3) promote potential solutions, and (4) increase communication among interested stakeholders.

METHODS

We proposed, coordinated, and co-chaired the symposium and panel discussion on “Managing Human-Wildlife Conflicts on the Hard Edges” at the 4th IWMC on July 11, 2012. The symposium was scheduled for 1 hour and 40 minutes and consisted of 4 experts who each made 25-minute presentations on various issues of the subject. Symposium speakers included: Michael J. Somers, Lecturer, Centre for Wildlife Management, University of Pretoria, South Africa; Edward E. Bangs, Coordinator (retired), United States Fish and Wildlife Service’s Western Gray Wolf Recovery Program, Montana, USA; Hanlie E. K. Winterbach, Research Scientist, Tau Consultants, Ltd., Botswana; and Christiaan W. Winterbach, Research Scientist, Tau Consultants, Ltd., Botswana. David L. Bergman provided comments that summarized the sessions. We recognize that an issue of this magnitude and scope cannot be fully addressed in a half-day session, but we hope that it serves as a catalyst for future discussions.

RESULTS

The symposium was informative and remarkably successful, with at least 150 people (over one-third of all Congress participants) in attendance. The presentations in this symposium addressed some of the most complex and controversial wildlife management issues of modern times: a summary of their comments are provided below.

Michael J. Somers presented on “Fences as Hard and Soft Edges for Wildlife.” His comments were based on years of research and a recent book titled “Fencing for Conservation” (Somers and Hayward 2012). Construction of fences has been widely sponsored and subsidized by governments in Southern Africa. Some people contend that conservation of biodiversity is not possible without protection that fences provide, while others argue that fences simply create zoos and restrict evolutionary potential. Four-strand fences are very “leaky” and do not restrict wildlife movements. Eight- to 10-foot woven-wire fences, that enclose up to 50% of the landscape in South Africa, impede movements, but initially were considered to provide conservation benefits because of the protection they provided wildlife from hunters and poachers. The probability of species decline is inversely proportional to the permeability of the fences. Greater concerns are associated with what is going on inside the fences: enetic manipulation and persecution of carnivores. He proposed “smart fences” that are species-specific, impermeable fences for protecting endemic species, and community-based fences.

Edward E. Bangs spoke on “Controversies and Complexities of Wolf Restoration in North America.” He provided research-based facts on population growth, mortality rates, dispersal distances, impacts on prey species, and pack persistence. Wolves were extirpated in the lower 48 states by 1930, reintroduced into Yellowstone National Park in 1995, and delisted (but challenged) in 2008. Wolves killed a documented 1,700 cattle, 3,300 sheep, 40 goats, 30 llamas, and 20 horses from 1995 to 2010. The numbers are small compared to livestock production figures in the area, but the impacts to individual producers are significant. Over 1,700 wolves were killed (nearly 15% of the total population) during the same period to address damage complaints. Nonlethal methods have worked to a limited extent and a compensation program paid out over $2 million U.S. for livestock losses. It is clear that the American public will continue to debate the relationships between large predators, private-public land, and human interests. Although science has served the profession well in many situations, we must recognize that it sometimes is a poor tool for resolving complex legal, political, and human conflicts associated with expansion of the range and population of gray wolves.
Hanlie E. K. Winterbach’s presentation was titled “Dynamics of Human-Carnivore Conflicts: Key Factors and Related Principles in the Conservation of Large African Carnivores.” She stated that populations of all 7 species of large African carnivores (LACs; lion, leopard, cheetah [Acinonyx jubatus], African wild dog [Lycaon pictus], spotted hyaena [Crocuta crocuta], striped hyaena [Hyaena hyaena], and brown hyaena [H. brunnea]), are declining and the primary problem is conflicts with humans. She modeled the key ecological, socio-economic, and political factors related to LAC conservation and human-wildlife conflicts (Winterbach et al. 2012). Key ecological factors include prey availability, livestock predation, interspecific competition, carnivore range, ecological resilience, wildlife diseases, and population viability. Key socio-economic factors include people’s attitudes and behavior, carnivore costs, and carnivore benefits. Key political factors include wildlife policy development and implementation, conservation strategies, and land use and zoning.

Christiaan W. Winterbach presented on “Dynamics of Human-Carnivore Conflicts: The Botswana Carnivore Landscape.” He identified the distributions of LACs in Botswana, based on 10 years of extensive aerial surveys and related them to human land-use practices, prey densities, and potential range expansion. Legal protection of LACs varies by species and across time, with some designated as game animals (spotted hyena), partially protected (lion), and protected (cheetah, African wild dog). Based on legal protection, the range of the lion corresponds well with the expected range, whereas ranges of cheetahs and African wild dogs are less than expected. Densities and distributions of LACs are shaped by key ecological and socio-economic factors. Planning, prioritization, and implementation of species- and site-specific conservation strategies are critical for the future of LACs.

DISCUSSION
The panel discussion included an audience of about 80 people and we believe was professionally beneficial to all participating. Scheduled to last only 20 minutes, discussion went on for 60 minutes before it was necessary to close the session. Generalized topics raised in questions and statements by the audience included: prioritization of conflicts (drought versus disease versus livestock predation); perceived versus real conflicts; identification of key drivers in human-wildlife conflicts; religion and economics as drivers of tolerance; attitudes and behaviors of preservationists versus conservationists; role of non-governmental organizations in education and changing attitudes; wildlife as publicly- versus privately-owned resources; removal of fences reduces spatial constraints and increases management options for elephants; cascading effects of fences on mesopredators; relocate people rather than wildlife; compensation increases tolerance of people for conflicts; hunting as a tool for resolving conflicts; acceptance of hunting increases with time and familiarity; predator control to promote game species; compensatory versus additive mortality; tolerance of dholes (Cuon alpinus) in India; management of nuisance and damage-causing baboons (Papio ursinus) in urban areas of South Africa; and Indian wolves (Canis lupus pallipes) causing depredation of livestock and attacking people outside of forested areas in India. Generalized responses by panel members included the following: management must be directed toward reducing damage to tolerable levels; include the human component in management; site- and species-specific management strategies; improved damage assessment; problem definition is a prerequisite to management; research in human dimensions is needed to understand tolerance levels for damage; an artifact of increasing predator populations is human-wildlife conflict; burden of predators is disproportionately passed on to a few; all benefits and costs of LACs must be considered; compensation increases tolerance for damage, but can be a “black hole;” hunting of predators can be socially acceptable; hunting engages the public in management; and management is necessary to avoid “frustration killing.”

David L. Bergman’s closing comments were based on notes he made during the sessions. He identified several cross-cutting themes from the speakers’ presentations:

- Carnivores are the resource managers of the landscape affecting other predators, prey, and subsequently habitat.
- Population levels of native prey often predicate the extent of conflicts.
- Minimum core areas for carnivores need to be designed and managed, based on the largest and widest ranging carnivores.
- Ecological resilience is a necessity to enable carnivore adaptation to human-modified landscapes.
- Each human has a unique value system based on life experiences and education.
- People’s behaviors, not attitudes, determine the impact on large carnivores.
- Decision-making often is impacted by fear, fairness, politics, and economics.
- Livestock protection needs to be socially acceptable and not just implementable.
- Farmers and landowners often are willing
to try alternative methods if cost-effective, provided for free, or they do not affect their bottom-line.

- Nonlethal techniques may not provide long-lasting solutions.
- Hunters will value predators if allowed to legally take animals without endangering population sustainability—thus predators become more valuable.
- Buy-in begins with providing locals with tools and techniques for survival.
- Modeling can help with buy-in if the right questions are defined and asked.
- In the industry of agriculture, a business model affects decision-making.
- Conflicts need to be minimized to allow for predator expansion and landowner tolerance.
- Conflict seems to be the current designer of ecology.
- Conflict mitigation is never-ending.
- Wildlife management seemingly has evolved into the science of controversy.
- If unchecked, controversy can destroy partnerships and good intentions.
- Conservation likely will fail where local impacts are not addressed—benefits versus costs to local individuals and communities are important, as needs must equal benefits.
- The need is great for integrated conservation strategies to resolve human-wildlife conflicts.

We believe that the symposium and panel discussion on “Managing Human-Wildlife Conflicts on the Hard Edges” at the 4th International Wildlife Management Congress in Durban, South Africa, on 11 July 2012 was popular, informative, and thought-provoking. Although we were not able to meet all of our goals, we hope that these sessions and this paper will serve as a catalyst for other similar sessions in the future that may lead to the resolution of human-wildlife conflicts for the betterment of people and wildlife at the hard edges.

ACKNOWLEDGMENTS
We appreciate the commitment of M. J. Sommers, E. E. Bangs, H. E. K. Winterbach, and C. W. Winterbach who presented and addressed questions during the sessions. We thank the audience members who participated in the panel discussion. Finally, we thank The Wildlife Society - Wildlife Damage Management Working Group, U.S. Department of the Interior - Fish and Wildlife Service, and Mississippi State University - Center for Resolving Human-Wildlife Conflicts for providing support for the sessions.

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THE EFFECT OF WOODLAND CARIBOU RANGE COMPONENTS ON HABITAT SELECTION AND FORESTRY ACTIVITY

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ABSTRACT: Woodland caribou (Rangifer tarandus caribou) across North America are in peril mainly due to forestry and the concomitant increase in predation; augmenting the need for sustainable management practices. Identifying and protecting core areas (CA) and important habitats can be used to reduce the impact of forestry. We determined how habitat selection and forestry activity differ between spatial components (i.e., different CA definitions and use intensities of the home range) and temporal components (i.e., seasonal CA of caribou). Objective CAs were defined using the Area Independent method and arbitrary cores were predefined using the 50% density contour. We used fixed-kernel density to create different spatial components for the calving season for 2 GPS-collared female caribou and objective CAs were created for each season (i.e., calving, post calving, rut and winter). Habitat preferences were assessed using the Manly selectivity index and the proportion of cutovers was used to infer the amount forestry. Mixed models were constructed for each CA, use intensity and season. We found that the arbitrary CA was half the size of the objective CA. Neither habitat preference nor the amount of cutovers changed between the spatial components while the proportion of cutovers varied across seasons. Bogs and coniferous forests had the highest selectivity indices in all range components. Our results suggest that the use of arbitrary cores may underestimate intensively used areas and that the seasonal components of range use should be considered when developing management plans.

KEYWORDS: core area, fixed-kernel home range, forestry, habitat selection, Newfoundland, range components, Rangifer tarandus caribou, woodland caribou.
Estimation of temporal components is necessary because habitat preferences, resource requirements, sensitivity to disturbances, and predation risk vary across seasons (Rettie and Messier 2000, O’Brien et al. 2006, Racey and Arsenault 2007, Briand et al. 2009, Hins et al. 2009). In spring for instance, predation is greater on caribou (Seip 1992, Trindale et al. 20) and their use of harvested landscapes to access green forage (Hins et al. 2009) increases their exposure to predators. Therefore, the combination of seasonal and range components of woodland caribou can be used to focus conservation management plans on important areas in each season.

Our study aims to assess the consequences of using different methods to delineate core areas and to determine important habitats and the amount of cutovers across different components of caribou ranges. Our objectives are to: (1) determine the space use patterns of caribou within their home range using an objective core area method; (2) determine how habitat selection and forestry activity differ with spatial components, (i.e., objective core, arbitrary core and home range), and use intensity levels; and (3) to demonstrate the effect of temporal components on habitat preferences. We assessed the space use patterns of the caribou and determined whether the objective core area differed from the arbitrary, predefined core area commonly used. We predicted that there will be stronger selection for refuges (e.g., bogs and coniferous forests) and lower preference for risky habitats (e.g., cutovers), and that the amount of cutovers will be lower in areas with higher use. We also expected seasonal variation in the selectivity indices of each habitat type and the amount of cutovers because vulnerability and life requirements vary between seasons.

STUDY AREA

The study took place in central Newfoundland within the forestry management zone 5 (49°N, 56°W) and covered approximately 6,737 km². The northern portion was bisected by the Trans-Canada Highway and contained the towns of Bishop’s Falls at the east and Millertown to the west (Figure 1). The topography of the area was characterized by flat to gently rolling landscape with many wet lowlands.

The major forest type was dense coniferous stands of mainly black spruce (Picea mariana) and some balsam fir (Abies balsamea) with sparse deciduous patches of trembling aspen (Populus tremuloides) and white birch (Betula papyrifera). Non-forested areas including wetlands and shrublands were also common. The summers are mild and wet (16°C) and the winters are cool (− 7°C) with an average snow accumulation exceeding 4 m per year (Mahoney and Virgl 2003). Logging operations, mainly clear cutting, have been ongoing since the 1920s in the study area focusing primarily on conifers for pulp and paper (Mahoney and Virgl 2003).

The woodland caribou in the study area were sedentary ecotypes that perform only small seasonal migrations (Bergerud 1971) and occurred in small groups of 5 to 30 individuals. Light hunting was still allowed in certain districts (Wildlife Division 20) despite the population declining in recent years (Trindale et al. 2011). Much of the study area was limited to human access, although some areas could be accessed by public and logging roads. Besides humans, predators of woodland caribou included lynx (Lynx canadensis), black bear (Ursus americanus; Bergerud 1971) and the recently arrived coyote (Canis latrans; Schaefer and Mahoney 2007). Wolves were historically the major predator on the woodland caribou on the island however they were extirpated in the 1920s (Bergerud 1971). The only other wild ungulate on the island was the introduced moose (Alces alces).

METHODS

Location data (1 location every 2 hours) from 12 GPS-collared (LOTEK 4400, Newmarket, Canada) female caribou from 5 different herds were used to create the seasonal core areas for 2008-2009. The caribou were captured by the crew members of the wildlife department of Newfoundland using stratified random sampling to allow collars to be more evenly distributed across the landscape and herds; each collared caribou represented a small herd of 5 to 30 individuals. Health Canada approved the capture and collaring protocol under experimental studies certificates 60021 and 60022. We used the woodland caribou reproductive season dates derived by the
Newfoundland and Labrador Wildlife Division to create the temporal components: including calving (20 May – 10 June 2008), post calving (1 July – 30 August), fall rut (1 September – 31 October), and winter (16 December 2008 – 31 March 2009) seasons. The seasons were divided by periods when the caribou are in migration or display major changes in behaviour (P. Saunders, Wildlife Division Newfoundland and Labrador Government, personal communication). Caribou locations (n = 21,858) were entered into the Geographical Information System for analysis. Of these locations, 89% (n = 19,375) were located within the study area.

The spatial components of the range (i.e., the different core areas and home range, hereafter called cores, and the use intensities) were created using fixed kernel density estimation in the animal movement v-2.04 BETA package in ArcView v-3.2 with cell size of 100 meters. We found least squares cross-validation for bandwidth selection inappropriate in this case because the core areas produced were conservative and fragmented. Therefore, to determine the bandwidth appropriate to construct the cores, we tested several bandwidths ranging from 400 to 1200 m. We concluded the 1000 m bandwidth produced the best cores for our purpose because it obscured the fine detail while highlighting the most prominent features of the range for most individuals.

The density contours used for the cores and use intensities were 50% (i.e., the most commonly used arbitrary contour; Laver and Kelly 2008), 75% (i.e., the average density contour across all seasons, estimated using the AIM; Powell 2000), and 95% - the home range (Table 1). An objective core area is obtained using the AIM by plotting the density contours versus their percentage of the home range. If a core area exists, a depression in the curve of the relationship would be observed indicating a clumped distribution. The point on the curve that is the greatest distance from a line with slope of negative 1 illustrates the area that is most intensively used and the density contour can be determined for the objective core area. The average density contour obtained using the AIM was 72% (SE = 1.04, n = 39). This contour was rounded up to the nearest 5% (75%) because of the precautionary principle and the animal movement program only creates isoclines in increments of 5 percent (Hooge and Eichenlaub 1998). These density contours were also used to create the 3 use intensity levels; low use (75 to 95% contour), medium use (50 to 75% contour), and high use (area within the 50% contour). Figure 1 illustrates the density contours used to create the cores and the use intensities for each caribou within the study area. The spatial components were only created for the calving season since it is an important season for woodland caribou and requires a better understanding of the critical habitats (Racey and Arsenault 2006). Only the objective contour was used when comparing across all 4 seasons.

We obtained digital vegetation coverage from Newfoundland’s Forest Service inventory database and classified the information into 9 general habitat categories (Table 2). The mean patch size was 3.8 ha ranging from 4,678 to less than 0.001 hectares. Data on the vegetation coverage was obtained from interpretation of aerial photos by the forestry department mainly during the years 2002, 2003, and 2004 and projected in MTM 2 (North American Datum of 1983) in a Geographic Information System. The land covered by municipalities and agricultural fields were omitted from the map since they covered less than 1.0% of the study area.

<table>
<thead>
<tr>
<th>Range characteristics</th>
<th>Mean</th>
<th>SD</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Objective core area (density contour)</td>
<td>71.34</td>
<td>6.30</td>
<td>55.00</td>
<td>85.00</td>
</tr>
<tr>
<td>Caribou locations in objective core (%)</td>
<td>75.04</td>
<td>9.81</td>
<td>52.29</td>
<td>89.77</td>
</tr>
<tr>
<td>Arbitrary core 50% (ha)</td>
<td>582.12</td>
<td>205.46</td>
<td>382.3</td>
<td>3,282.5</td>
</tr>
<tr>
<td>Objective core area 75% (ha)</td>
<td>1,258.11</td>
<td>484.85</td>
<td>746.37</td>
<td>2,443.79</td>
</tr>
<tr>
<td>Home range 95% (ha)</td>
<td>3,424.35</td>
<td>1,513.90</td>
<td>1,292.67</td>
<td>7,286.35</td>
</tr>
<tr>
<td>Use intensities</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>50% isopleth (ha)</td>
<td>508.86</td>
<td>228.22</td>
<td>402.94</td>
<td>863.22</td>
</tr>
<tr>
<td>50-75% isopleth (ha)</td>
<td>786.55</td>
<td>442.25</td>
<td>349.59</td>
<td>1,640.57</td>
</tr>
<tr>
<td>75-95% isopleth (ha)</td>
<td>2,695.75</td>
<td>1,472.47</td>
<td>1,380.11</td>
<td>4,842.56</td>
</tr>
</tbody>
</table>

Table 1. Spatial and temporal range characteristics of the woodland caribou (n = 12) in central Newfoundland, Canada, in 2008.

<table>
<thead>
<tr>
<th>Habitat type</th>
<th>Description</th>
<th>Total area (ha)</th>
<th>Proportional area (%)</th>
<th>Validation score (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Barren</td>
<td>Rock, soil, and barren land</td>
<td>105</td>
<td>2</td>
<td>85</td>
</tr>
<tr>
<td>Bog</td>
<td>Bog, wet bog, wetland and other wetlands</td>
<td>1,149</td>
<td>18</td>
<td>100</td>
</tr>
<tr>
<td>Coniferous</td>
<td>Coniferous trees planted 75-97% of the total area</td>
<td>274</td>
<td>6</td>
<td>100</td>
</tr>
<tr>
<td>Deciduous</td>
<td>Deciduous trees planted 75-97% of the total area</td>
<td>47</td>
<td>2</td>
<td>100</td>
</tr>
<tr>
<td>Disturbed</td>
<td>Forest fire, wind damaged, fixed damaged, insect mortality</td>
<td>322</td>
<td>3</td>
<td>100</td>
</tr>
<tr>
<td>Mixed forest</td>
<td>Not define</td>
<td>227</td>
<td>3</td>
<td>89</td>
</tr>
<tr>
<td>Shrub</td>
<td>Hardwood and shrub with few trees</td>
<td>1,322</td>
<td>29</td>
<td>72</td>
</tr>
<tr>
<td>Water</td>
<td>Lakes and major rivers</td>
<td>558</td>
<td>8</td>
<td>100</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>6,463</td>
<td>100</td>
<td></td>
</tr>
</tbody>
</table>

Table 2. The area (ha), proportional area (%) and validation scores (%) of the habitat categories available to woodland caribou in central Newfoundland, Canada in 2008.

*Validation score with disturbed habitat omitted.
*We were unable to correctly identify any disturbed forests because the habitat type resembled other habitats.
We tested the accuracy of the habitat map by photointerpretation of 225 random points on aerial photographs from 2003 and 2004 (Hansen et al. 2001). The map accuracy was 78.9% (Table 2). The accuracy would increase to 87.8% if the disturbed habitat type was omitted. The disturbed habitat type was difficult to distinguish from other habitats because it represented areas disturbed by fire, wind, flood or insect damage and resembled other habitat categories. However, considering the accuracy of the other habitat categories, we trust that the disturbed habitats were correctly defined and are reliable indicators of recent and historical natural disturbances.

We assessed habitat selection within the range of the caribou or the second-order level (Johnson 1980). At this scale, the available habitats are those within the study area and the used habitats are those within the range components of the caribou. This scale allows us to determine the amount of refuges and risky landscapes available to the caribou and the proportion of cutovers was used to infer the amount forestry. We conducted the habitat selection at this scale because selection is driven by the most important factor influencing fitness (Rettie and Messier 2000). In addition, the composition of the home range is known to affect caribou survival and reproduction (Wittmer et al. 2007, McCarthy et al. 2011). We determined the habitat composition in each of the range components by creating Manly’s standardized selectivity indices (b) for each habitat type (Manly et al. 2002, Mahoney and Virgl 2003). This method allows us to determine the selection patterns within the caribou ranges across the available landscape and it tolerates the exclusion of habitats not used within any of the core or use intensities but that are found within the study area (Manly et al. 2002, Mahoney and Virgl 2003).

Using the information from the habitat map, 2 stand age groups, recent cutovers (dating from 1999 to 2008) and regenerating cutovers (dating from 1960 to 1998), were used to depict the amount forestry activity within the cores and use intensities and for each season. The total area of cutovers dating from 1960 to 2008 in the study area was 37,351 ha, 8,120 ha of which were regenerating cutovers and 29,231 ha recent cutovers. The average patch size of recent cutovers was .3 ha ranging from 0.0086 to less than 0.00 hectares and regenerating cutovers was 1.5 ha ranging from 95.3 to 0.0086 ha. The proportion of cutovers was compared to determine if the set selectivity indices (b) differed between each habitat type. We also conducted mixed models for each habitat type and cutover category using the selectivity indices or proportion of cutovers as the dependent variable and each of the spatial or temporal components as the independent variable. This allowed us to determine if the habitat selectivity indices and proportion of cutovers differed between the spatial and temporal components. We used the post-hoc pairwise comparisons test, to compare each of the selectivity indices or proportions and we controlled for type I error using a Tukey adjustment. We used 5% as the significance level for each test. The assumptions of the models were assessed using residuals plots.

**RESULTS**

**Spatial Components**

The average density contour from the objective method was 71.5% (SE = 1.04, n = 39) and ranged from 55 to 85% (Table 1). The average sizes for the spatial components were 582 ha (SE = 32.9, n = 39) for the arbitrary core, 1,258 ha (SE = 77.6, n = 39) for the objective core, and 3,424 ha (SE = 242.4, n = 39) for the home range. The objective core was approximately 2 times greater than the arbitrary core and 3 times smaller than the home range. The average size of the objective core for the calving season was 555 ha (SE = 164.0, n = 11; 0.082% of the study area) with a maximum of 803 and minimum of 382 hectares.

The use of different habitat types was non-random for the 50% core (F8, 80 = 8.40, P ≤ 0.001), 75% (F8, 80 = 10.46, P ≤ 0.001) and the 95% (F8, 80 = 15.08, P ≤ 0.001) home range. Bogs, coniferous forests, shrubs, and water bodies had a higher selectivity index than mixed and disturbed forests in all 3 spatial components, (all P < 0.05; Figure 2).

Differential use of habitat types was apparent in the high (F8, 80 = 8.40, P ≤ 0.001), medium (F8, 80 = 9.81, P ≤ 0.001), and low (F8, 80 = 13.54, P ≤ 0.001) use intensity levels. In this case only bogs and coniferous forests consistently had higher selectivity indices than mixed, disturbed and deciduous forests (all P < 0.05; Figure 3). Only 1 caribou had deciduous forests within the high use area, 2 caribou in the medium use area, and 3 caribou in the low use area.

There was no evidence that the preference for each habitat type changed across the cores or use intensities (all P > 0.05; Table 3). Similarly, the proportion of recent cutovers did not vary across the cores (F2, 8 =
0.17, \( P = 0.846 \) or use intensities (\( F_{2,7} = 1.02, P = 0.408 \)) and nor did the older cutovers for either cores (\( F_{2,8} = 2.71, P = 0.126 \)) or use intensities (\( F_{2,8} = 2.78, P = 0.121 \); Figure 4). When all cutovers were summed together, the proportion of cutovers still did not vary across the cores (\( F_{2,8} = 1.04, P = 0.397 \)) or use intensities (\( F_{2,8} = 3.39, P = 0.086 \)). One caribou had much greater proportion of cutovers compared to the others with a total of 23, 18 and 26% for the high, medium and low use areas respectively.

**Temporal Components**

The use of the different habitat types was non-random during the calving \( (F_{1,80} = 10.46, P \leq 0.001) \), post calving, fall rut, and winter \( (F_{1,80} = 4.99, P = 0.001) \), fall rut \( (F_{1,80} = 4.9, P \leq 0.001) \), and winter \( (F_{1,80} = 5.3, P \leq 0.001) \) seasons. The habitat types in calving, post calving, and fall rut seasons had the same ranking pattern as with the spatial components. Cutover were less preferred than bogs in both calving and winter \( (P > 0.05) \); however, during all other seasons mixed forests had a lower bi than bogs.

The only selectivity indices that differed between seasons were those for cutovers \( (t_{2,24} = 2.82, P = 0.044) \), post calving \( (t_{2,24} = 3.45, P = 0.011) \), and fall \( (t_{2,24} = 3.71, P = 0.005) \) than the winter season. For mixed forests, only the selectivity indices of calving \( (t_{2,24} = -3.18, P = 0.020) \) and regenerating \( (t_{2,24} = 0.75, P = 0.542) \) cutovers did not differ between seasons; however, when they were summed together a difference between seasons was observed \( (F_{3,12} = 4.01, P = 0.034) \). The proportion of all cutovers for both fall \( (t_{12} = 3.31, P = 0.028) \) and post calving \( (t_{12} = 2.98, P = 0.049) \) were greater than in winter.

**DISCUSSION**

Applying an arbitrary rule for the assignment of a
core area can misidentify the areas of intensive use, and jeopardize management and conservation efforts. Therefore, an objective method such as the AIM method is recommended to increase the precision of estimates of animal space use patterns, which can in turn influence the areas prioritized for protection (Powell 2000, Laver and Kelly 2008). Indeed, we found that core areas estimated with the AIM method had an area twice as large as those estimated with the arbitrary method.

The caribou in this study intensively used a large portion of their home range possibly reflecting their highly mobile nature. The application of the arbitrary core to identify lands for conservation may reduce the area required by woodland caribou to satisfy their physiological needs and hinder conservation efforts by inadvertently including anthropogenic disturbances within the intensively used areas potentially affecting mortality (Wittmer et al. 2007, McCarthy et al. 2011).

In our study, the spatial components of the home range did not influence the habitat selection patterns; bogs and coniferous forests were preferred over cutovers, deciduous and mixed forests, a pattern commonly observed in other studies (Bradshaw et al. 1995, Mahoney and Virgl 2003, Schaefer and Mahoney 2007, Courtois et al. 2008, Hins et al. 2009). The high preference for bogs and coniferous forests allows for refuge from predation and supports the anti-predator strategy that caribou display in other systems (Wittmer et al. 2007, Hins et al. 2009). In addition, they support an abundance of forage (Bradshaw et al. 1995) making them crucial habitats for caribou.

Contrary to our prediction, the selectivity indices did not differ across any of the spatial components of the home range. The presence of core areas and similarity of habitat preferences across the use intensity levels may indicate that the caribou select...
for finer scale habitat characteristics not captured by this study (Rettie and Messier 2000, Johnson et al. 2001, Racey and Arsenault 2007, Briand et al. 2009). Also, habitats may have been spatially autocorrelated within home ranges thwarting detection of any change in preferences (Legendre 1993). However, studies using other methods of capturing habitat selection such as frequency of habitat use did show a change in habitat preference across different spatial components (Mosnier et al. 2003, Hins et al. 2009). This suggests that the habitat composition may not mirror the actual use patterns of the habitat types. The caribou in our study may show a change in preference patterns across spatial components using different techniques, such as frequency of use in each habitat type.

Similar to other studies (Rettie and Messier 2000, Mahoney and Virgl 2003, Mosnier et al. 2003, Hins et al. 2009), habitat preference did change among seasons. For example, in winter caribou showed a high preference for coniferous forests, mixed forests and bogs and a stronger avoidance of cutovers. The avoidance of cutovers in winter could be a response to higher predation and higher snow accumulation (Rettie and Messier 2000, Smith et al. 2000, Hins et al. 2009). Snow accumulation is likely the larger factor responsible for this avoidance since the high snow accumulation (> 4m) would likely reduce the ability to access forage. In addition, the caribou’s primary winter predator, the coyote, avoids cutovers in winter because of the accumulated snow that hamper movements (Thibault and Ouellet 2005). Also, the calving season had a much stronger selection pattern than the other seasons. Calves are most vulnerable during this season (Trindale et al. 2011) and thus it was only in the calving season when both bogs and coniferous forests were preferred over cutovers.

The similar proportions of cutovers between use intensities was surprising because of the general avoidance of cutovers observed in other studies (Smith et al. 2000, Mahoney and Virgl 2003, Courtois et al. 2008). However, the use of recent and regenerating cutovers have been observed in other studies (Briand et al. 2009, Hins et al. 2009) which may be a consequence of habituation to cutovers or fidelity to historical ranges and not the selection for cutovers (Rettie and Messier 1998, Smith et al. 2000). Each of the use intensity levels for 1 caribou in the study had approximately 20% cutovers. The inclusion of cutovers may be because they are perceived as lower risk from the absence of wolves and as beneficial because of available food sources (Russell et al. 1993, Briand et al. 2009) and avoidance of insect pests (Graham 1992). However, cutovers are also commonly found in the core areas of coyotes (Boisjoly et al. 2010), a significant predator

**Figure 5.** Selectivity indices of habitat types (Deci. = deciduous forests, Dist. = disturbed forests) by woodland caribou in central Newfoundland, Canada, in 2008 for the calving (a), post calving (b), fall rut (c), and winter (d) seasons. Variables sharing the same letter are not significantly different based on the Tukey test. Error bars represent the SE.
on caribou in Newfoundland possibly creating an ecological trap. However, little information exists on the impact of cutovers on caribou predation by coyotes in Newfoundland.

**MANAGEMENT IMPLICATIONS**

Managers should require the use of objective core area estimators for determining important caribou habitats to protect because arbitrary techniques were found to be too small and failed to identify key areas. The inclusion of cutovers within the core areas of woodland caribou is concerning because of the potential for higher predation possibly creating an ecological trap.

Pooling all seasons for the construction of core areas may reduce the effectiveness of determining important seasonal areas because the length of each season can differ, biasing the location of the core areas. Thus, we recommend constructing core areas for each season to fully capture the different habitat components, thus allowing managers to identify and focus management efforts within different seasonal periods.

**ACKNOWLEDGMENTS**

We thank the field staff of the department of Environment and Conservation Wildlife Division and Department of Natural Resources for collecting the data. Financial support was provided by the Government of Newfoundland and Labrador and the Fonds de Recherche du Québec - Nature et Technologie. Many thanks to G. Body, J. Couture, S. Engelhardt, J. Grant, A. Hamelin, J. Jaeger, and N. Melnycky for their support and helpful comments.

**LITERATURE CITED**


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DEVELOPING AN ORGANIZATIONAL RELOCATION POLICY

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ABSTRACT: Whether it is called relocation, translocation, assisted migration, or even assisted colonization, numerous programs have proposed to move animals and plants from one place to another. Climate change is now a prominent driver of such plans. As organizational oversight policy was being reviewed, this study examined the recent history of such relocations to see if lessons could be derived that are applicable to the development of new proposals. Reviewing published reports of >400 relocations, plus earlier reviews of specific groups, several relevant concerns emerged. Many animal and plant species that were the focus of the plans were threatened or endangered at the state, national, or international level, or are at risk of being so listed. To alleviate the threats to these species, it was often suggested that remaining individuals be moved from places where they still survive to another site where they may flourish. Yet <50% of the results show that the individuals maintained themselves at the new sites. I found insufficient understanding of the many variables involved in relocation. Even releases of common game species showed numerous failures to establish. Another frequent omission was the failure to create sound plans to manage and monitor the relocated animals. Reports of relocation success were often related to the reasons underlying the relocation: to rescue a doomed population, to re-establish an extirpated population, to extend the range, to create a harvestable population, or to supplement an existing population. Special concerns may relate to the possible effect to the source population (e.g., which life stage [adult, juvenile, egg] is being relocated). I examined aspects of translocation with respect to the efficacy of these practices for many species at risk, and how that efficacy may contribute to the development of policy guidance for the agency.

KEY WORDS: head starting, introduction, policy development, re-introduction, relocation, species at risk, threatened species, translocation.


Translocation (or relocation) is defined by the International Union for the Conservation of Nature and Natural Resources (IUCN), as the capture, transport and release or introduction of species from 1 location to another (IUCN 1987). Game birds and fishes are commonly introduced for sporting purposes. When considered in the light of support for species at risk, translocation as a tool is often proposed to reduce the risk of a catastrophe to a species with a single population, to aid the natural recovery of a species or re-establish a species where barriers might prevent it from doing so naturally. In the context of defense installations and possible defense policy development, the logic is that individuals could be relocated to a place where they would face fewer stressful encounters with military activity. This sounds like a good idea, but what precautions need to be taken?

The objective of this study is the examination of the nature and level of concern in planning and evaluating relocation/translocation of species in the military installation context. Special attention was given to its application to species at risk, which are of conservation concern but may not yet be officially recognized as candidates for listing under the Endangered Species Act. The whole topic of moving wildlife from place to place has numerous underlying motives, and a correspondingly large number of terms that have been applied to the different aspects of the process. The IUCN lists several terms associated with the practice, defining translocation as the “movement of living organisms from one area with free release in another” (IUCN 1987). Although there are nuanced differences, I use relocation and translocation here as synonyms.

Can translocation do good?

Basically, translocations of non-game species, and especially of species at risk, are attempts to ensure the safety and ultimate survival of a species. If successful, this may often be a good solution in highly disturbed landscapes where barriers stop natural colonization from occurring. Especially when the causes for the original loss are known, and have been remediated, the reintroduction of a species to its former habitat may have significant ecosystem values.

Is there potential for harm?

First, there is a large risk that the translocation will fail. I ask if that is acceptable for a species-at-risk. For example, the relocated animal may suffer physiological consequences of translocation, such as stress, causing a weakened immune system. Introduced animals may carry known or undetected disease which then may spread to the existing populations. The new site may not support the species due to life history requirements which are unknown. At worst, harm may be done to both the donor and the recipient popula-
tions. If viable offspring are removed from a location where they are likely to survive, then that population has lost all or a portion of an age cohort which could contribute to its viability.

METHODS
This study was restricted to a survey from the published literature of reports of studies which moved a species from 1 location to another. Thus, “relocation,” “translocation,” and “re-introduction” when used as search terms retrieved >350 potentially relevant publications. All taxa were included, and the survey may be considered random for use of any of those terms. Several reports examined, or re-examined, previous reports of relocations, so that >400 individual sets of project results were represented. Reports were examined with the following questions in mind: (1) What was the purpose of the relocation? (2) What life stages were used? (3) How many individuals were released? (4) What was the source? (5) Was the relocation a success? (6) How was this determined? 7) If a failure, was a reason presented? Not all reports covered each of these elements, however, so a full matrix could not be developed.

RESULTS
Previous Reviews
Several surveys of the efficacy of translocation used different methodology and included different sets of studies. Griffith et al. (1989) surveyed 81 conservation organizations and 2 follow-up surveys focused on either threatened, endangered, or sensitive (TES) species or native game species. Broadly, they found that success for game species was >80%, but for TES species only 44%. Dodd and Seigel (1991) reviewed 25 published reports of relocation actions involving amphibians and reptiles. Their overall conclusion was that only 5 of these projects were fully successful, through confirmation that a self-sustaining population had resulted. This proportion (20%) is about half the value reported by Griffith et al. (1989) for birds and mammals.

Fischer and Lindenmayer (2000) evaluated 180 examples reported in 124 articles in conservation-oriented publications that concentrated on 116 classified as reintroductions, where almost equal numbers were considered successful (26%) as unsuccessful (27%). Success for 47% was undetermined. Where cause of decline was known, but not corrected, no reintroductions were known to be successful. Even where it was corrected, 7% were clear failures. The survey by Germano and Bishop (2009) covered 25 studies of amphibians and 39 of reptiles. They determined that 42% of 85 new introduction projects could be considered successful, while 28% failed to establish a self-sustaining population. For 29%, success was not determined. They found no significant difference in success between reptile and amphibian translocations. Translocations of reptiles undertaken for purposes of removing animal-human conflicts were mostly undetermined or clear failures. Animals taken from the wild were more likely to succeed in their colonization, and the number of animals released was important. Projects that released >1,000 individuals (e.g., amphibians) were twice as likely to succeed as those releasing ≤100.

In her follow-up of the survey by Griffith et al. (1989), Wolf et al. (1996) generally supported the results from Griffith et al. (1989). Respondents were asked to provide their “…professional understanding as to the cause(s) of the outcome…” This represented a slightly different approach to determining results, because opinions may well identify very different causality than can be demonstrated statistically. She identified 19 factors reported to be influential in success or failure. One interesting conclusion was that human disturbance or interference appeared to be very minor factors in the failed or declining populations reported to the study.

Dodd and Seigel (1991) presented some examples of successful and unsuccessful projects known to them, and made 2 observations. First, why, in spite of there not being many verifiable successes, are relocation proposals so popular? They identified several underlying motives, both positive and negative. The first was that it is often good publicity for a biologist or manager to propose relocation, especially if humane considerations are at work, such as in moving animals from a development site. Local and regional politics enter here as well, because development pressures may be irresistible, and delays create extreme pressure on the state to allow relocation as an alternative to destruction. Another set of interrelated motives revolve around the fact that some translocations are successful, and that many more are perceived to be successful, based perhaps on inadequate follow-up monitoring or sampling. The lack of clear reasons for failure enters here as well, because slow decline may mask failure in the absence of total population loss. Second, Dodd and Seigel (1991) made a series of recommendations which they felt biologists and others proposing relocation actions should take into consideration: (1) know the cause for original decline; (2) know the biological constraints; (3) understand population genetics and social structure; (4) be sensitive to issues of possible disease transmission; and (5) don’t overlook need for long-term monitoring.

Fischer and Lindenmayer (2000) examined success as related to the original reason for the relocation. The categories used were: (1) to solve human-animal
interaction problems; (2) to supplement game populations; and (3) for conservation purposes. Responses for different purposes were very different. The relocation of nuisance animals had the least likelihood of success. Because these studies are often evaluated based on whether or not the nuisance animal returns to its original habitat, the death of the animal at its new location may appear to be a success. Fischer and Lindenmayer (2000) suggested methods to improve the success of conservation-motivated relocations. There were 5, paralleling to some degree those of Dodd and Seigel (99). With slight rewording, they were: () use an appropriate approach to accomplish the aims desired; (2) define success more clearly; (3) continue monitoring to clearly determine success or failure; (4) keep better track of true cost; (5) publish the results, even if the action fails.

Current Study
In addition to the reviews noted above, I identified 101 mostly recent published papers which reported in some manner on the results of relocation studies. Not all reports were specifically focused on the fact of the relocation, per se. Some focused on aspects such as reproduction of the species involved, or the reasons for failure in later years. The papers included both direct and indirect data. In 1 “indirect” study, Germano and Bishop (2009) reviewed 91 translocations covering 25 amphibian and 39 reptile species and reported 2 types of results, the first being the overall success rate, and the second the reasons for failure, where known. I did not include this report in the 101 studies examined, because I could not identify which species fell in which category.

The majority of the papers, however, considered 1 species, or a group of related species which could be categorized for at least some of the evaluation factors noted above. In a few cases however there were larger numbers of projects wrapped into a publication (e.g., 70 attempts at introduction reported in 1 case). When the number of studies was summed, the papers included here represent the results of 419 introduction attempts. Figure 1 shows the different taxa covered in the entire dataset.

When one examines these results, several different conclusions may be drawn from the different factors tracked. For varying reasons, I could not track some of the elements mentioned above. The life stage, for example, was omitted because certain characteristics applied totally (or largely) to only 1 group of species. The other factors included do represent items where I feel real comparisons may be made. The factors I recorded were species, continent, country, whether it was a reproducing population, the author’s claim of success, number of projects, reason for failure (if applicable), reason for study, and citation to original publication.

I found that the totals for a few factors did not add to 101. Some reports showed results as “mixed”; thus, the number of successes plus failures was <101. Finer discrimination is possible for many of the elements for the 101 species dataset. Figure 2 shows the distribution of taxa within that dataset as compared to those in the 419 entry group. The larger set included
proportionally fewer mammals and more invertebrates. Some other groups, such as birds, fish, and reptiles were present in almost the same proportions. Because the number of samples was 101, the numbers displayed are almost identical to percentages for Figure 2.

Defining “success”, I applied the sole criterion of whether or not a reproducing population was established as a result of the activity. Using these criteria, I found that slightly fewer than half the studies had been successful (Figure 3). Within the text, however, authors claimed 61% of the studies were successful (Figure 4). Why the difference? Apparently, the failures were about equivalent, but the authors saw far fewer studies whose outcome was unknown. They may also have redefined “success” in terms of adjunct objectives rather than the one I used.

DISCUSSION
Following this review of hundreds of reports of studies involving relocation of many species within many taxonomic groups, I conclude that relocation should not be undertaken as a simplistic action with assured results. It would appear that, for at-risk taxa, the likelihood of success in a relocation effort is <50%. Considering that the studies reported were typically undertaken with the oversight of institutions of higher learning, it may be that ad hoc relocations undertaken by on-site personnel can be expected to be even less successful in most cases. I identified several good review studies of relocations, some dealing with only selected taxa and others more general. Several of these studies (e.g., Dodd and Seigel 1991, Fischer and Lindenmayer 2000) analyzed reasons for failure and developed sets of recommendations for future studies. The recommendations of both of these studies are included in this report, and may represent a minimum set of requirements for future undertakings.

Finally, the IUCN position statement Translocation of Living Organisms (IUCN 1987), while >20 years old, provides extremely relevant general advice on what to do and not do when undertaking a translocation. Virtually every study that I reviewed which was not successful could have attributed the failure to lack of accordance with ≥1 of the recommendations in the IUCN document. That said, some of the successful ones probably did not follow rigorously every recommendation, and were successful in spite of those omissions.

MANAGEMENT IMPLICATIONS
The primary recommendation arising from this study is that development of official policy and a review procedure within the agency, aligned with the existing requirements of existing environmental regulations would be highly desirable. It is clear that formulae for success have not yet been proven, and failures are frequent. Inadequate long-term follow up of these activities is all too common. As a result, I believe that the general or widespread application of translocation as a means to sustain Species at Risk is probably not in an agency’s best interests, although some specific cases may be acceptably well-documented. Whether or not formal operational guidance has been prepared, it is highly encouraged to become familiar with the IUCN relocation recommendations. They serve to alert personnel to the major concerns from the biological and ecological per-
spective, although they are not oriented to meet concerns which may arise while following individual agency procedures.

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LITERATURE CITED

Bertschinger et al. • Immunocontraception of African Elephants

PORCINE ZONA PELLUCIDA
IMMUNOCONTRACEPTION OF AFRICAN ELEPHANTS (LOXODONTA AFRICANA):
BEYOND THE EXPERIMENTAL STAGE

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ABSTRACT: In southern Africa there is a need for elephant (Loxodonta africana) population control, especially in small- to medium-sized, fenced reserves. The objectives of this study were to investigate the effects of porcine zona pellucida-immunocontraception on the reproductive rate as well as the safety during pregnancy of elephant cows in 7 private game reserves in South Africa. A total of 108 individually-identified cows were treated and monitored for 6 years. Primary vaccinations consisted of 400 or 600 μg porcine zona proteins with 0.5 ml Freund’s modified complete adjuvant and boosters of 400 or 200 μg zona proteins with 0.5 ml Freund’s incomplete adjuvant. Vaccine was delivered remotely: year 1, primary plus 2 boosters 3—6 weeks apart; year 2 onwards, annual boosters. Birth of calves was monitored continually and the result expressed as a percentage of cows treated on an annual basis. During years 1 and 2, 35 (32.4%) and 22 (20.4%) calves were born, respectively. No more calves were born from year 3 onwards. One cow conceived around the time of primary vaccination and a second between the primary vaccination and first booster. Two calves died soon after birth from unrelated causes. The remainder survived and were normal healthy calves. One hundred percent of cows passed the 4-year, 67.6% the 5-year, and 47.2% the 6-year inter-calving interval. The results show that it is possible to achieve a contraceptive efficacy of 100% in small- to medium-sized free-ranging populations of African elephants.

KEY WORDS: African elephants, game reserves, efficacy, immunocontraception, porcine zona pellucida, vaccine.


According to Kerley et al. (2008) the impact of African elephants (Loxodonta africana) on ecosystems and biodiversity is difficult to assess. Elephants improve conditions for other herbivores while negatively affecting a number of other animals. They decrease the diversity of plant species on the one hand while improving the landscape on the other. Be that as it may, the general consensus amongst reserve managers is that elephant populations, left uncontrolled in small- to medium-sized reserves, will have a negative impact on habitat and thus biodiversity of the reserve concerned. Small reserves are defined as around 100 km² and medium-sized reserves around 500 km² (Mackey et al. 2006). In South Africa many elephant populations were introduced into smaller fenced parks during the 1980s and 1990s. Previously maximum annual population growth rates were estimated at 4—7% (Hanks and McIntosh 1973, Calef 1988) whereas recently they have been found to be >10% (Mackey et al. 2006). The rapid population increase known as irruptive growth (Mackey et al. 2009) from density-independent population increase, may eventually lead to die-offs from starvation (Caughley 1970). The need to manage elephants, while controversial in the Kruger National Park (KNP), is well-accepted in small- to medium-sized fenced reserves (Mackey et al. 2006). Traditionally culling has been regarded as the method of choice for controlling large populations (Slotow et al. 2008). However, besides the opposition from many quarters, culling is hardly applicable to smaller populations. The practice is to cull entire breeding herds to avoid stress of family members left alive (Slotow et al. 2008). In practice this probably seldom happens. Also, in small populations this could mean removing all, half or a third of the breeding animals, depend-
ing on the size of the population. Despite being very costly, translocation is regarded as an ideal solution; however, in South Africa, habitat availability is limited (Delsink et al. 2006).

Besides enlargement of parks the only other option to manage elephants is to decrease reproductive success by means of contraception. In selecting a contraceptive method for free-ranging mammals such as African elephants, it must be efficient, reversible, safe, remotely deliverable, which largely determines the cost and have a minimal impact on the social behaviour of the target species (Kirkpatrick and Turner 1991). Immunocontraception using porcine zona pel- lucida (pZP) vaccine satisfies all these requirements as has been shown in intensive studies in domestic and wild horses (Liu et al. 1989, Kirkpatrick and Turner 2008) white tailed deer (Turner et al. 1992, McShea et al. 1997, Rutberg and Naugle 2008) and a number of other free-ranging and captive-held herbivores (Dei- gert et al. 2003, Frank et al. 2005, Kirkpatrick and Frank 2005, Kirkpatrick et al. 2009). The putative mechanism for the success of pZP immunocontraception is the production of antibodies that bind to ZP proteins of target animals’ oocytes to prevent sperm binding (specifically to ZP3; Clarke and Dell 2006), fertilisation and thus pregnancy. Fortunately zona proteins have been well-conserved across mammal species and antibodies to pZP have been shown to recognise the African elephant ZP proteins (Fayrer-Hosken et al. 1999).

Earlier immunocontraception trials on African elephants in the KNP showed that the porcine pZP vaccine is safe and effective as a contraceptive in African elephant cows and, in the short term, reversible (Fayrer-Hosken et al. 1997, 1999, 2000). The final efficacy rate achieved was 80% of vaccinated cows. This initial work was followed by an extensive study in the Greater Makalali Private Game Reserve (Makalali). The vaccine was shown to be 100% effective and, once all cows pregnant at inception of the program had calved, no more calves were born from the third year of the project (Delsink et al. 2006, 2007).

This paper describes the effect of pZP vaccine on reproductive rate of free-ranging African elephant cows in medium and 6 small reserves over periods of 6 years. Makalali is included in this study as additional information is included.

**STUDY AREA**

The game reserves, their sizes, provincial locations in South Africa and the broad vegetation types of each are shown in Table 1.

**METHODS**

The protocols for this project were approved by the University of Pretoria’s Animal Care and Use Committee, Project number: VO49/11. The elephants on each of the 7 reserves were introduced by means of translocation and adult bulls were present on each reserve. Game reserve, year of inception of the contraception program and number of cows of reproductive age (Laws 1966, Lee et al. 1995) vaccinated during year 1 were: Makalali, 2000, 18 cows (Delsink et al. 2006, 2007); Mabula, 2002, 4 cows; Phinda, 2004, 19

<table>
<thead>
<tr>
<th>Game Reserves</th>
<th>Makalali</th>
<th>Mabula</th>
<th>Phinda</th>
<th>Shambula</th>
<th>Thornybush</th>
<th>Welgevonden</th>
<th>Kaingo</th>
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</thead>
<tbody>
<tr>
<td>Size</td>
<td>24 500 ha</td>
<td>8 000 ha</td>
<td>22 800 ha</td>
<td>8 000 ha</td>
<td>11 548 ha</td>
<td>35 000 ha</td>
<td>8 461 ha</td>
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<tr>
<td>Provalional location</td>
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<td>Limpopo</td>
<td>KwaZulu Natal</td>
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<td>Limpopo</td>
<td>Limpopo</td>
</tr>
<tr>
<td>Broad vegetation type</td>
<td>Granite lowveld</td>
<td>Central sandy bushveld</td>
<td>Zululand lowveld/West Maputaland clay bushveld</td>
<td>Central sandy bushveld</td>
<td>Granite lowveld</td>
<td>Waterberg mountain bushveld</td>
<td>Central sandy bushveld/western sandy bushveld</td>
</tr>
<tr>
<td>Population size (n) year 1</td>
<td>53</td>
<td>11</td>
<td>92</td>
<td>10</td>
<td>35</td>
<td>117</td>
<td>9</td>
</tr>
<tr>
<td>Cows treated (n) year 1</td>
<td>23(^b)</td>
<td>4</td>
<td>19</td>
<td>4(^c)</td>
<td>19</td>
<td>35</td>
<td>35</td>
</tr>
<tr>
<td>Age of cows year 1 (years)</td>
<td>12-50</td>
<td>13-16</td>
<td>10-35</td>
<td>19-25</td>
<td>6-31</td>
<td>9-44</td>
<td>10-40</td>
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<tr>
<td>Cows (n) calved before treatment</td>
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<td>3</td>
<td>18</td>
<td>No data</td>
<td>11</td>
<td>25</td>
<td>No data</td>
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<tr>
<td>Estimated mean monthly calving% before treatment (number of years)</td>
<td>21.7(^e)</td>
<td>25.0% (3)</td>
<td>21.0% (6)</td>
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<td>16.7% (6)</td>
<td>20.6% (6)</td>
<td>No data</td>
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<tr>
<td>Mean annual calving% during years 1 and 2 of the study</td>
<td>32.6%</td>
<td>12.5%</td>
<td>39.5%</td>
<td>25.0%</td>
<td>15.8%</td>
<td>30.0%</td>
<td>25%</td>
</tr>
</tbody>
</table>

\(^a\)Classifications as per McNab and Rutherford (2010)

\(^b\)18 cows were treated in 2000 (Delsink et al. 2000); 2 added in 2001 and 3 in 2002

\(^c\)Only vaccinated twice during year 1 (2004) and boosted for another 3 years. Moved to another game reserve in 2008 with no bulls

\(^d\)Per number of cows judged to be of breeding age

\(^e\)Adapted from Delsink et al. 2006
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cows; Shambala, 2004, 4 cows; Thornybush, 2005, 19 cows; Welgevonden, 2005, 35 cows and Kaingo, 2005, 4 cows. Additional cows were added during years 2 (2 cows, Makalali) and 3 (3 cows Makalali) (Delsink et al. 2006, 2007), and in years 4 (Mabula, \( n = 1 \)) and 5 (Makalali, \( n = 5 \)) cows were removed from the program so that they could be allowed to reverse. Either before or during the course of year 1 each target animal was individually identified (Delsink et al. 2002). This allowed vaccination to take place on an individual cow basis. Prior to treatment, the populations typically had an inter-calving interval of 4.5—5 years and at inception of each program cows were at various unknown stages of reproduction. At the beginning of year 5, after 4 years of contraception, the Shambala population was captured and translocated to Entabeni Private Game Reserve where no bulls of reproductive age were present. The vegetation type (Waterberg) is similar in the 2 reserves. Monitoring of the cows continued on the new reserve.

**Vaccine and Vaccine Delivery**

The pZP antigen was produced by a modification of the methods described by Dunbar et al. (1980). The vaccine was manufactured at the Science and Conservation Centre, ZooMontana, Billings, MT for the 2000—2003 vaccinations. Thereafter, it was produced and supplied by the pZP Laboratory of the Department of Production Animal Studies, University of Pretoria. During year 1 each cow of reproductive age was given 3 pZP vaccinations; primary of 400 \( \mu g \) (600 \( \mu g \) at Makalali and Mabula) pZP in 1 ml phosphate buffered saline (PBS) with 0.5 ml Freund’s complete modified adjuvant (Sigma Chemicals Co., St Louis, MO); 2 boosters of 200 \( \mu g \) (400 \( \mu g \) at Makalali and Mabula) pZP each in 1 ml PBS with 0.5 ml Freund’s incomplete adjuvant (Sigma Chemicals Co., St Louis, MO). The intervals between vaccinations were 3—6 weeks. The 4 cows each in Shambala and Kaingo only received 1 booster during year 1. This was followed by annual boosters with 200 \( \mu g \) (400 \( \mu g \) at Makalali and Mabula during 2000—2003; thereafter 200 \( \mu g \) pZP in 1 ml PBS with 0.5 ml Freund’s incomplete adjuvant. Shortly before use, the pZP antigen and adjuvant were mixed using 2 syringes joined by means of a connector. The fluid was pushed forwards and backwards between the syringes approximately 60 times creating a stable emulsion. Darts were then loaded forwards and backwards between the syringes approximately 60 times creating a stable emulsion. Darts were then loaded with the emulsion. During the first 3 years at Makalali, Dan-Inject® (DAN-INJECT ApS, Børkop, Denmark) darts with 60 mm needles were used (Delsink et al., 2007). Thereafter and on the other reserves, Pneu-Dart® (Pneudart, Williamsport, PA) darts with 50 mm 13 gauge needles with gel collars were used. Elephants were either darted from the ground or a helicopter. To facilitate the identification of cows within a group already darted during helicopter work, most cows were vaccinated with Pneu-Dart® mark and inject darts containing a pink dye (Wonder Mark®, Mafuta Products, Ventersdorp, South Africa).

**Monitoring of Cows Post Vaccination**

Cows on all game reserves were mostly seen 1—3 times a week but during wet periods spotting intervals were sometimes longer and as much as 2 weeks between sightings. Birth dates of new calves were taken as the date of first sighting. Mothers were identified with their calves that were either in close proximity or being nursed (Delsink et al. 2002). Duration of gestation was taken as 22 months (Laws 1966, Hodges et al. 1994). Using this period, stage of gestation could be calculated in cows pregnant at the time of inception of contraception or shortly thereafter. To simplify reporting, gestation was divided into trimesters as follows: first trimester, 0—8 months; second trimester, 9—15 months and third trimester, 16—22 months.

**Data Analysis**

The total number of calves born per annum for years 1 through 6 was expressed as a percentage of the total number of cows treated each year. Expressing the an-
Annual reproductive rate as a calving percentage (calves born/annum/100 cows) was preferred to population growth rate because of the varying circumstances of each population. The \( \chi^2 \) test was used to analyse annual differences in calving percentage. As the year of commencement of contraception differed between reserves (2000—2005) they were normalised so that the date of primary vaccination was the first day and 365 days later the last day of year 1. The cows added to the trial during years 2 and 3 at Makalali were also normalised to fit the data. Day 366 was then the start of year 2 and so on. The numbers of cows treated each year varied from 98—108 as a result of some individuals being removed from the program for reversal and with the translocation of 4 cows from Shambala to Entabeni Game Reserve (Table 2). With a gestation period of 22 months pregnancies that would have been initiated during the first 4 years at the old reserve would have given rise to the birth of calves during the first 2 years at Entabeni.

**RESULTS**

Approximate calving data was available for 5 of the 7 reserves prior to inception of contraception and varied from 16.7% to 25.0% in terms of annual calving percentage per cow of breeding age (Table 1). The mean calving percentages for years 1 and 2 of the trial varied from 12.5% to 39.5% between reserves with an overall annual mean of 26.4% for 108 cows. This was equivalent to 1.06 calves per cow per cycle of 4 years and a mean inter-calving interval of 3.8 years.

Following primary vaccination 35 calves were born during year 1 and 22 during year 2 providing calving percentages of 32.4% and 20.4%, respectively (Table 2). No calves were born during years 3, 4, 5 and 6 (\( P < 0.001 \)). With the exception of 2, all calves (\( n = 57 \)) were conceived prior to the primary vaccination (Table 3). One calf was conceived around the time of primary vaccination and the other between the primary vaccination and the first booster. Of the 108 cows vaccinated during year 1, 100% passed the 4-year, 67.6% (73 of 108) the 5-year and 47.2% (51 of 108) the 6-year intercalving interval. From the calving dates it was apparent that 57 cows were at various stages of pregnancy. One calf died as a result of a physical injury soon after birth and another as a result of haemorrhage from the umbilicus at birth. The remaining calves were healthy and survived. Table 3 indicates the stage of pregnancy when the calves as embryos or foetuses were exposed to the primary vaccination. There was an even spread of gestation stages in terms of pregnancy trimesters. About one third (\( n = 18 \)) were in the first trimester and were thus exposed to possible effects of the vaccine as early as the embryonic stage.

Calculated according to their calving dates, 2 of the 5 cows that were allowed to reverse during year 5 at Makalali, (last vaccination in June 2004) conceived 23 and 34 months after the last treatment, respectively. The remaining 3 cows at Makalali and 1 cow at Mabula have yet to calve 7 years after the cessation of treatment.

**DISCUSSION**

The mean calving percentage of 26.4% for all 7 reserves during years 1 and 2 of the trial was higher than those recorded prior to inception of contraception in the 5 reserves that had historical data. There are 2 possible reasons for these differences. Firstly, contrary to post-inception, birth dates of calves were not available in most reserves during the previous years and ages of calves were estimated according to shoulder height (Laws 1966, Jachmann 1988, Lee and Moss 1995). Secondly, a number of cows in the trial only reached reproductive age around year 1 of the trial and some were even younger. Although we compensated cow numbers to correct for this, figures quoted prior to inception of the program should only be regarded as estimates. The mean calving percentages for years 1 and 2, on the other hand, are in agreement with recently published data for introduced populations (Mackey et al. 2006) which quotes population growth rates of up to and even exceeding 10%. Our data for years 1, 2 and the mean for the 2 years shows population growth rates of 10.8%, 6.1% and 8.5%, respectively. The fact that fewer calves were born during year 2 than year 1 is likely to be due to chance.

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**Table 3.** Calves born after the primary vaccination showing the stage of gestation and conception in relation to primary vaccination presuming a gestation period of 22 months for African elephant cows (*Loxodonta africana*) (Hodges et al. 1994).

<table>
<thead>
<tr>
<th>Trimester of gestation at time of primary vaccination</th>
<th>Number of calves</th>
<th>Conception in relation to primary vaccination</th>
<th>Number of calves</th>
</tr>
</thead>
<tbody>
<tr>
<td>First trimester</td>
<td>18</td>
<td>Before</td>
<td>55</td>
</tr>
<tr>
<td>Second trimester</td>
<td>20</td>
<td>Around the primary</td>
<td>1</td>
</tr>
<tr>
<td>Third trimester</td>
<td>19</td>
<td>Between primary and 1st booster</td>
<td>1</td>
</tr>
</tbody>
</table>

By the year of conception presuming a gestation period of 22 months, 57 cows were at various stages of pregnancy. One calf died as a result of a physical injury soon after birth and another as a result of haemorrhage from the umbilicus at birth. The remaining calves were healthy and survived.
As reported previously (Delsink et al. 2006) no more calves were born from the third year onwards in this study. The mean inter-calving interval of <4 years (3.8 years) has been passed in 100% of females while 67.6% and 47.2% of cows passed 5- and 6- year intervals, respectively. For Makalali and Mabula, excluding the cows taken off contraception, no calves born to the original 21 treated cows after 10 years. These data once again demonstrate the efficacy of the vaccine to control fertility in African elephant.

The question that surely must be asked is, from when onwards in terms of the initial vaccinations are elephant cows infertile? Our data reflect that 1 cow conceived around the time of the primary vaccination when the antibody titre was either baseline or just starting to increase. A second cow conceived between the primary vaccination and first booster indicating that at least one booster is necessary to provide sufficient antibodies to block sperm-zona binding and thus a pregnancy from taking place. All remaining 55 cows that calved after inception of the program had conceived prior to the primary vaccination. The elephants that were treated with a lower dose of pZP (400 μg, primary and 200 μg for boosters vs 600 μg, primary and 400 μg for boosters) gave equivalent results from year 3 onwards. They have, however, not been treated for as long as the cows in Makalali and Mabula. Based on these results we have routinely used the lower dosage regimen since the beginning of 2004. The doses required to achieve immunocontraception with pZP in the elephant are considerably smaller than is required for horses (100 μg) if one adjusts for body mass. Similarly the dose of GnRH vaccine used to immuno-regulate androgen secretion in the pig (400 μg) is relatively much larger than is used for the same purpose in African elephant bulls (600 μg; Denys et al. 2010).

Curiously, the 95% efficacy of pZP immunocontraception achieved over a period of 17 years in wild horses (Kirkpatrick and Turner 2008) was lower than that achieved in African elephant cows. The collective efficacy of pZP immunocontraception in 24 ungulate species, 25 bears and 11 sea lions was 93.3% and ranged from 60% (nyala; Taurotragus angasi) to 100% in 16 other species such as bison (Bison bison), mountain goats (Oreamnos americanus), wapiti (Cervus canadensis), fallow deer (Dama dama) and moose (Alces alces; Frank et al. 2005). Efficacies within the ungulate species varied from 60—83% in 6 species and 91.6—100% in the remaining 18 species. All animals reported by Frank et al. (2005) were held and treated in zoos. The 1 major advantage that possibly contributes to the success rate in elephants is the long interval of approximately 4 years between calves. This means that, with a gestation period of 22 months, the elephant cow takes approximately 2 years to conceive again. The precise physiology of the latter period is unknown but thought to be similar to lactation anoestrus seen in some domestic species like the sheep and the pig (Bertschinger et al. 2008). Ahlers et al. (2012) in a 1-year study found that of 9 adult and 5 subadult cows treated with pZP and monitored by means of faecal progesterone metabolite concentrations, 6 showed regular and 2 irregular luteal cycles. Three cows that showed no proper luteal cycles had calved a mean of 9.3 months and 21.3 prior to the start and end of the study, respectively. This would indicate that the cows were acyclic or in anoestrus throughout the study period. The remaining 3 acyclic cows were subadults indicating that they had not reached puberty yet. Furthermore, at any 1 time, one can expect approximately 50% of African elephant cows to be pregnant (Bertschinger et al. 2008). Thus in the elephant there is ample time during the presumed anoestrus and pregnancy periods to achieve good pZP antibody titres capable of preventing fertilisation and pregnancy later on. The very first 2 pZP-immunocontraception field trials in elephants recorded contraceptive success rates of only 56% and 80%, respectively (Fayrer-Hosken et al. 2000). In both trials 400 μg and 200 μg pZP was used for the primary and booster vaccinations, respectively, but instead of Freund’s adjuvants, synthetic trehalose dicorynomycolate (5 mg per vaccinations) was used as adjuvant. During the first trial (n = 18; efficacy 56%) the boosters were administered 6 weeks and 6 months after the primary vaccination. In the second trial (n = 10; efficacy 80%) 2 booster were administered at 2-weekly intervals.

Just like as in the previous study in elephants (Delsink et al. 2006), we clearly demonstrated the safety of pZP-immunocontraception during pregnancy. The loss of 2 out of 57 calves was accidental and unrelated to the use of the vaccine. Irrespective of the stage of pregnancy during vaccination, the 55 other calves were born healthy and viable and have survived until today. This means that no developmental abnormalities during pregnancy could be attributed to the use of the vaccine in elephants.

Previously short-term reversibility of pZP-immunocontraception could be demonstrated in 3 free-ranging African elephant cows after only year 1 treatment (primary and 2 booster vaccinations). Twenty-two months after the primary vaccination all 3 cows were found to be pregnant on transrectal ultrasound examination (Fayrer-Hosken et al. 2000). The present study investigated the reversal potential in 6 cows that had been treated for 3 (n = 1) and 4 (n = 5) years, respectively. Two cows treated for 4 years conceived 25 and 36 months after the last treatment with pZP. The
remaining 4 cows have yet to produce a calf. It seems thus that the interval from last treatment to reversal is quite variable. Two studies have investigated ovarian function using faecal progestagen metabolite concentrations in free-ranging elephant cows treated for 2 to 3 (n = 14; Ahlers et al. 2012) and 4 years (n = 4; Benavides et al. 2012), respectively. With exception of cows that had calved a mean of 9.3 months before the start of the study (Ahlers et al. 2012), all adult females treated with pZP showed evidence of luteal activity. Although the studies were 12 and 14 months long, respectively, neither one could demonstrate negative effects of pZP treatment on luteal ovarian function. In both studies, faecal progestagen concentrations were significantly lower during the dry than the wet season and, in pZP-treated and untreated cows at Entabeni (Benavides et al. 2012), seasonal anoestrus was common. Importantly the latter study showed that, in the absence of conception, free-ranging elephant cows do not necessarily cycle continuously as was previously believed (Bertschinger et al. 2008).

MANAGEMENT IMPLICATIONS

Immuonococontraception using the pZP vaccine is highly effective as a method of birth control in African elephants. Calving in treated animals ceases two years after inception of the program. It is 100% safe for conceptuses at any stage of development. The delivery of the vaccine is remote and at no stage requires target animals to be caught or immobilized. The largest population treated so far is Welgevonden with 117 elephants of which 35 cows of reproductive age were targeted. Despite the mountainous terrain of the reserve, a 100% efficacy was achieved meaning that the treatment of larger populations is feasible. pZP-immunocontraception presents a proactive means of population control in elephants whereas culling is reactive, and once implemented, must continue indefinitely if it is to succeed. Reproductive rate in African elephants is density dependent (Laws 1969; Laws et al. 1975) and the response to culling will be an increase in this rate. Finally, pZP-immunocontraception provides managers of small to medium-sized reserves with a viable and ethically acceptable means of controlling reproduction in African elephants.

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