2012

Sorta Situ: The New Reality of Management Conditions for Wildlife Populations in the Absence of "Wild" Spaces

Barbara A. Wolfe
Wildlife and Conservation Medicine

Roberto F. Aguilar
Humane Society of the United States

A. Alonso Aguirre
George Mason University

Glenn H. Olsen
U.S. Geological Survey Patuxent Wildlife Research Center

Evan S. Blumer
OsoMono, LTD, Gahanna, Ohio and Retired Executive Director

Published in *New Directions in Conservation Medicine: Applied Cases in Ecological Health,* edited by A. Alonso Aguirre, Richard S. Ostfield, and Peter Daszak (New York: Oxford University Press, 2012).

Authors

Barbara A. Wolfe, DY.M., Ph.D., Dipl. A.C.Z.M.
Director
WIldlife and Conservation Medicine
The WIlds, Cumberland, Ohio

Roberto F. Aguilar, D.V.M.
Veterinarian
Cape Wildlife Center
Humane Society of the United States
Bamstable, Massachusetts

A. Alonso Aguirre, D.V.M., M.S., Ph.D.
Associate Professor
Department of Environmental Science and Policy
George Mason University
Executive Director
Smithsonian-Mason School of Conservation
Front Royal, Virginia

Glenn H. Olsen, D.V.M., Ph.D.
Veterinary Medical Officer
U.S. Geological Survey Patuxent Wildlife Research Center
Laurel, Maryland

Evan S Blumer, V.M.D., M.S.
Principal
OsoMono, LTD, Gahanna, Ohio
and
Retired Executive Director
The Wilds, Cumberland, Ohio

40

SORTA SITU

The New Reality of Management Conditions for Wildlife Populations in the Absence of "Wild" Spaces

Barbara A. Wolfe, Roberto F. Aguilar, A. Alonso Aguirre, Glenn H. Olsen, and Evan S Blumer

The rate of species loss today is approaching catastrophic levels. Scientists project that over the next two decades, more than 1 million species of plants and animals will become extinct. E.O. Wilson has estimated that "the rate of loss may exceed 50,000 a year, 137 a day . . . this rate, while horrendous, is actually the minimal estimate, based on the species/area relationship alone" (Kellert and Wilson 1993, p. 16; Aguirre 2009). Ever-expanding communities, strained natural resources, changes in land use, and other anthropogenic drivers are compromising ecosystems and rapidly changing the landscape and the availability of "wild" spaces.

One outcome of these changes is the manifestation of a new global reality for wildlife. Where truly "wild" populations are increasingly rare and more animals are managed in protected zones, refuges, and conservation centers, the difference between *in situ* (wild populations in native habitat) and *ex situ* (captive populations in non-native habitat) becomes less distinct (Fig. 40.1). In fact, most wildlife populations of today and tomorrow exist on a continuum between *in situ* and *ex situ*. We define this new reality as *sorta situ*[1]—neither one nor the other—to describe the changing nature of population management in the 21st century (Aguirre and Pearl 2004). This chapter will use current examples to illustrate *sorta situ* populations and circumstances, and discuss considerations that are necessarily becoming part of the strategy of conservation medicine practitioners in managing and caring for wildlife populations in this changing global paradigm.

The continuum of conditions for *sorta situ* populations can be viewed across two key variables: available habitat, including space, habitat quality, and the maintenance of ecosystem processes; and management intensity, including healthcare and protection from outside threats. In the past, wildlife populations lived on large landscapes in their native habitat, without human intervention. Diseases and populations were, for the most part, self-limiting, and terrestrial animals were free to move in response to seasonal and

1 Term coined by John Jensen (2001), Environmental Program Director of the George Gund Foundation, describing "sort of" *situ* as an *ex situ* management strategy approximating *in situ* conditions.

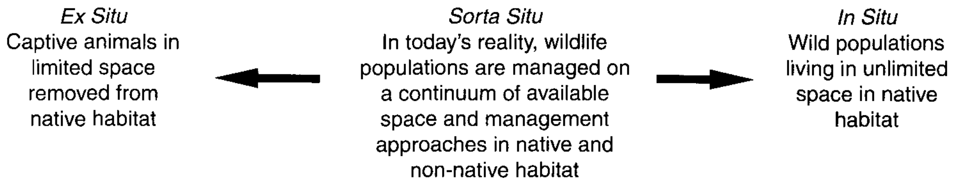

Figure 40.1:
The changing nature of wildlife management and conservation from *in situ* to *ex situ*, with *sorta situ* in between.

dietary needs. On the other end of the spectrum, zoos managed animals outside their native habitat in small captive groups, treating disease and injury on an individual basis and controlling nutritional input and reproductive output. Today, fences, borders, and human habitation limit the spaces wild populations can occupy, and small, fragmented populations require careful monitoring and management to avoid devastating population declines and extinction.

This new reality calls for a new approach to conservation medicine and management: a combination of *ex situ* developed skills, including small population management, practical veterinary care, and intensive reproductive management, linked to landscape-scale monitoring and field skills that include habitat restoration, reintroduction techniques, epidemiology, ecological modeling, behavioral ecology, and community-based conservation.

SPECTRUM OF MANAGEMENT INTENSITY IN *SORTA SITU* POPULATIONS

Intensive Management of Wild Populations

Increasingly, "wild" populations live in limited spaces of varying sizes in their native habitat and, in some cases, are closely monitored and often visited by wildlife health professionals when necessary. The Mountain Gorilla Veterinary Project of Rwanda, Uganda, and the Democratic Republic of Congo is an example of this type of *sorta situ* model (Cranfield et al. 2002). In this program, the approximately 740 remaining "wild" mountain gorillas (*Gorilla gorilla beringei*) live in fragments of their native habitat, are visited frequently by humans through ecotourism, and are monitored closely by a team of veterinarians and staff, who respond to each individual gorilla's need for medical care. A majority of "wild" rhinoceroses in Africa similarly live in refuges and parks under armed guard. They are observed frequently, treated for injuries, and translocated in response to poaching and other risks.

Even in geographically unrestricted bird populations, individual veterinary care is being employed for population management. The California condor (*Gymnogyps californianus*) reintroduction program in Vermillion Cliffs, Arizona, California, and Baja California has used veterinary support for lead-poisoned birds since the problem was first detected in the late 1990s. Bullet fragments in carcass remains left by hunters and eaten by condors have markedly increased the number of chronically lead-poisoned birds in recent years. Condors are tested and treated in the field if possible, while severely lead-poisoned birds with crop stasis or other complications are admitted to zoo hospitals for treatment. Once treated, the birds are sent back to the original capture site for release and monitoring.

Whooping cranes (*Grus americana*), both the existing wild flock (migrating between Wood Buffalo National Park, Alberta, Canada, and Aransas National Wildlife Refuge, Texas) and two introduced flocks (a non-migratory flock in central Florida and an eastern migratory flock between Wisconsin and Florida), are monitored and medically managed by a transdisciplinary team belonging to many institutions. This team collaborates to rescue and rehabilitate any sick or injured whooping cranes. Most individual members of the introduced flocks are fitted with radio transmitters and benefit from frequent and intense monitoring by dedicated teams of wildlife biologists.

When a problem is detected or when routine captures are planned, veterinarians participate to examine the individual crane and obtain diagnostic samples, providing information on the health of both the individual and the population.

Limited Management of Captive Populations

While some "wild" populations receive intensive management and care, some "captive" populations thrive far from native habitat on expansive tracts of land, receiving little management or medical care. Such is the case in many game ranches, wildlife parks, and hunting preserves. In Texas, for instance, over 250,000 exotic ungulates live on game ranches (Teer et al. 1993; Mungall and Sheffield 1994; Demarais et al. 1998). Many of these herds are virtually unmanaged, and free-ranging populations of axis deer (*Axis axis*), fallow deer (*Dama dama*), sika deer (*Cervus nippon*), blackbuck antelope (*Antilope cervicapra*), nilgai antelope (*Boselaphus tragocamelus*), aoudad (*Ammotragus lervia*), and other exotic ungulates have been established through escape and release (Huerta-Patricio et al. 2005).

POPULATIONS ON THE SPECTRUM OF AVAILABLE SPACE

From Zoos to Conservation Centers

According to the Conservation Breeding Specialists' Group, there is a need for conservation of threatened species to be shifted from zoos to larger breeding and conservation facilities (CBSG Newsletter Jan. 2010). In such facilities, animals given more space and kept in larger, more natural groupings might exhibit more natural behaviors (Clubb and Mason 2003; Li et al. 2007) and experience reduced stress, improved health, and enhanced breeding success. A recent study found, in fact, that large enclosure size and the opportunity to interact with conspecifics was associated with reproductive success in the southern white rhinoceros, *Ceratotherium simum* (Metrione and Harder 2009).

As zoological institutions become more involved in *in situ* conservation efforts, a new emphasis has been placed on the challenges of current breeding and captive management paradigms and the creation of larger conservation facilities. As established by the recent strategic plan of the Association of Zoos and Aquariums, captive facilities "will ensure the sustainability of diverse wildlife collections in accredited zoos and aquariums; advance high standards of wildlife-focused animal care and welfare; and foster outcome-based conservation by connecting zoos and aquariums to the wild" (http://www.aza.org/StrategicPlan/). A recently formed consortium of such facilities in the United States known as the Conservation Centers for Species Survival comprises five of the largest land-holding institutions: Smithsonian Conservation Biology Institute in Front Royal, Virginia; White Oak Conservation Center in Yulee, Florida; The Wilds in Cumberland, Ohio; Fossil Rim Wildlife Center in Glen Rose, Texas; and the San Diego Safari Park in Escondido, California. This consortium combines research, management, and training efforts to improve conservation of animal species and natural resources. The combined space available for captive animal management in these five institutions is over 20,000 acres, and the consortium is dedicated to cooperatively studying how landscape-scale settings and new techniques in captive management will uniquely benefit certain wild animal populations.

Wild Game Farms, Ranches, and Preserves

Wildlife and exotic game are farmed and hunted on privately owned lands ranging from less than 100 to over 100,000 acres in many countries. Hudson et al. (1989) described three different management approaches—farming, ranching, and herding—used by managers of native and exotic ungulates. Ranched and farmed populations are both confined by fences, but at different levels of management intensity. While farming involves intensive genetic, medical, and nutritional management of the captive population, ranched animals are managed primarily as confined wild populations. Herding, the least intensive approach and the one used more often on game preserves, describes management of a wild population relying on natural migration patterns to move and control animals. For instance, Rocky Mountain elk (*Cervus elaphus*) and American bison (*Bison bison*) in the Greater Yellowstone Area of the northern United States are managed by herding between seasonal feeding grounds on public and private lands.

Protected Areas

With species extinction rates threatening to increase to nearly 1,000-fold background rates, establishing protected areas for wildlife management is considered the primary defense against extinction (Joppa et al. 2008). To date, approximately 60,000 protected areas, defined as "[An] area of land and/or sea especially dedicated to the protection and maintenance of biological diversity and of natural and associated cultural resources, and managed through legal or other effective means" (IUCN 1994), cover approximately 13% of the earth's land surface (Phillips 2003). Available land, ecosystem stability, and management strategies within these protected areas, however, may determine species success in protected areas (Chape et al. 2005). Surveys of fragmented forests have shown that forests of less than 100 km^2 are insufficient to protect small populations of vulnerable birds over long periods of time (Brooks et al. 1999; Ferraz et al. 2003). Strong inverse relationships have been demonstrated between reserve size and extinction rates (Newmark 1996, 2008), and between ecosystem stability and species diversity (Dobson 2009). In some of these cases, certain specialized management techniques, such as the release of captive-bred animals, are being employed to augment fragile populations of single species. However, management of these *sorta situ* environments will likely require more systematic, ecosystem-wide conservation practices for the maintenance of ecological balance.

Fences, Walls, and Borders

Fences are used for many, and sometimes combined, reasons (Bode and Wintle 2009). Veterinary fencing in southern Africa was historically intended to control the spread of animal pathogens such as foot and mouth disease, which is transmitted from wildlife reservoir hosts to domestic livestock with devastating economic consequences. Many of the wildlife reservoirs of these diseases crossed international borders and intermingled with susceptible stock, creating regulatory difficulties in managing both disease spread and food safety in farmed animals. Veterinary fences provided an anthropogenic disease barrier, easing regional political strain and decreasing disease spread between wild ungulates and livestock. However, fencing and isolation of populations have been found to be detrimental to free-ranging species and ecosystem processes by preventing seasonal migration in large mammals and decreasing species abundance and genetic diversity of wild populations (Bolger et al. 2008; Chase and Griffin 2009), as well as affecting human welfare by excluding populations from their traditional lands and natural resources (Western 2002; Hoole and Berkes 2010). In Scotland, where fences are maintained to protect forest habitat from destruction by deer, capercaillie (*Tetrao urogallis*) and black grouse (*Tetrao tetrix*) have experienced up to 32% annual mortality due to fence collision (Catt et al. 1994). The walls being built between Israel and Palestine and more recently along the U.S.–Mexico border can similarly affect terrestrial populations of wildlife by dividing conservation areas and refuges (Cohn 2007; Sayre and Night 2009; Flesch et al. 2010; Wildlife Society 2010). Disruption of transboundary movement corridors by impermeable fencing has isolated some wildlife populations on both sides of the border.

Clearly, fences and borders, compounding the reduction of wild spaces available to maintain population stability and genetic dispersal, affect connectivity and spatial distribution of populations. Without the application of scrupulous mitigation strategies (Flesch et al. 2010), the result of these barriers is further division and isolation of wild species into subpopulations, increasing the chance that they will require some level of management to survive in the *sorta situ* future.

Transfrontier Conservation Areas

By far the largest examples in the spectrum of available space are transfrontier conservation areas (TFCAs), recently established in Africa. TFCAs are a cooperative natural resource management strategy spanning numerous parks, reserves, and countries, encompassing over 1.2 million km^2 of land and providing vast expanses for wildlife. While innovative, TFCAs can be politically and economically challenging from a disease transmission and food safety perspective (Cumming et al. 2007). For protected areas to succeed in conserving wildlife and habitat, they must benefit the neighboring human populations (Phillips 2003; Hoole and Berkes 2010). Their planning and management must involve the local people, including social, economic, and conservation objectives, with long-term goals and political considerations in mind.

HEALTH CONSIDERATIONS IN THE MANAGEMENT OF *SORTA SITU* POPULATIONS

From intensively managed in unrestricted space to unmanaged in large enclosures in captivity, *sorta situ* populations encounter novel conditions, whether in the form of frequent human exposure, foreign climates, new food sources, new parasites and diseases, new competitors and predators, or even novel soil and substrates. As conservation medicine practitioners, we are now challenged to identify, predict, and assess the health impacts of the host of habitat changes that these populations encounter across the range of management scenarios, and to better manage these changes to conserve both wild populations and their habitats.

The Human–Wildlife–Domestic Animal Interface

Wildlife populations, regardless of their available space, no longer exist in isolation from humans, domestic animals, and their evolving diseases. Decreasing space and increasing globalization have led to an increase in emerging infectious diseases (Daszak et al. 2000). An ever-increasing human population and decreasing wildlife habitat, in combination with changing animal ecology and climatic conditions, are thought to have led to the emergence of diseases such as Lyme disease, Hendra and Nipah viruses, and hantavirus pulmonary syndrome (Daszak and Cunningham 2002). Animals maintained on game ranches in their native habitat can present a risk to their wild counterparts, due to movement and fence line contact. For instance, the recent spread of chronic wasting disease in wild deer (*Odocoileus* spp.) and elk (*Cervus elaphus*) in the northern United States most likely occurred by transportation of farmed cervids between states and across the U.S.–Canada border by the owners and employees of farming operations (Williams and Miller 2002). With the introduction of deer farming in New Zealand, the presence of malignant catarrhal fever in domestic sheep resulted in outbreaks of the disease in susceptible deer (Wilson 2002).

In southern Africa, ecotourism plays a bigger economic role than agriculture, forestry, and fisheries combined, leading to greater exposure of wildlife populations to human presence (Osofsky et al. 2008). Particularly in developing countries, human populations, and their associated domestic animals, are relatively dense at the borders of protected areas for various reasons (Kalema-Zikusoka 2005). In such areas, emerging and re-emerging zoonotic diseases are of persistent and increasing concern to the health of the wildlife, domestic animals, and humans.

The health problems observed in free-ranging wildlife populations today resemble those seen historically in captive wildlife. Where diseases once existed in ecological "balance" and were self-limiting on a population scale, fragmented populations now are more vulnerable to stress, reproductive suppression, decreasing genetic diversity, malnutrition, and environmental pollutants, all leading to reduced immune protection against disease. The result is that wild animals are more vulnerable than ever to the possibility that a disease could wipe out a local population.

Animal Density and Translocation: Effects on Disease Risk

In captive populations, common diseases have been managed by exclusion, quarantine, sanitation, and vaccination, while outbreaks tend to be more devastating to larger, less intensively managed populations. The management of native and non-native wildlife species on limited habitat is altering the ecology of pathogens. For instance, the introduction or reintroduction of predators to an area alters the behavior of prey animals, presenting a host of risks to the prey, including decreased foraging and reproduction, increased dispersal, and, potentially, increased spread of disease (Heithaus et al. 2009; Sih et al. 2010). This phenomenon has been studied intensively in the reintroduction of grey wolves (*Canis lupus*) to the Greater Yellowstone Ecosystem (Fortin et al. 2005; Creel and Christianson 2008, 2009). Often, indigenous wildlife harbor diseases to which they are resistant or are unaffected carriers, and transmit them readily to susceptible exotic species (Chomel et al. 2005). Such is the case with the parasite *Parelaphostrongylus tenuis*, which is carried asymptomatically by white-tailed deer in the northeastern United States and has caused lethal central nervous system parasitic migration in a yet incompletely defined number of species of domestic and exotic ruminants and camelids (Nagy 2005).

Animal density can be influenced in both unrestricted spaces and in captivity. Where supplemental feeding of wildlife occurs, diseases like brucellosis in Rocky Mountain elk—which normally calve in seclusion, limiting transmission of *Brucella abortus*—are spread more rapidly due to increased animal density and contact (Cross et al. 2010). Following the spillover of bovine tuberculosis to white-tailed deer in the northern United States, supplemental feeding of deer led to widespread tuberculosis in wild deer populations. White-tailed deer became a reservoir for tuberculosis, leading to spillback and increased incidence of tuberculosis in domestic cattle (Schmitt et al. 1997).

Anthropogenic changes in animal density can have unexpected effects on disease prevalence. Increasingly, ungulate herds are managed as either semi-free-ranging or free-ranging populations with supplemental feeding. In such circumstances, predictive models of transmission and susceptibility to disease are altered. For example, in a study of infection by multiple pathogens in red deer (*Cervus elaphus*) under different management conditions, host body condition was negatively associated with infection with the lungworm *Elaphostrongylus cervi*, as would be expected due to the availability of host resource for use in parasite defense. Conversely, host density was also negatively associated with parasite counts (Vicente et al. 2007). Epidemiological models predict that host population density would correlate positively with abundance of nematode parasites because high population density, and therefore high contact rates, would increase the potential for transmission under natural conditions (Arneberg 2001). In this case, however, population density is artificially manipulated by supplemental feeding, and therefore the animals near feeding stations, although at high density, have improved body condition and therefore improved immune defenses. This is not the case in captive and semi-free-ranging populations in which all animals have access to feed, but are fed and sheltered at a small number of stations, where aggregation will encourage parasite transmission.

Augmenting Wild Populations Using Captive Management Techniques

Maintaining species genetic diversity is becoming increasingly difficult as fewer and more fragmented "wild" populations exist. Captive populations have been managed in zoological institutions for decades as safety nets or "arks" for supporting wild populations, for repopulating lost populations, and as a management system for sustaining genetic diversity both in the wild and in captivity (Soulé et al. 1986; Lees and Wilcken 2009). Augmentation of isolated wild populations through captive breeding may be fundamental, at least in the short term, to the survival of some wild populations (Conde et al. 2011). This is the case for the Mexican wolf (*Canis lupus baileyi*), the black-footed ferret (*Mustela nigripes*; Box 40.1), the kakapo (*Strigops habroptila*), the sandhill crane (*Grus canadensis*), and whooping cranes at present.

There are an estimated 47 wild Mexican wolves in the Blue Range Reintroduction Area between Arizona and New Mexico. Released wolves have been illegally hunted to the point at which the wild population is not sustainable without intense monitoring and protection. To augment the wild population, seven founder Mexican wolves held in zoos and conservation centers have produced approximately 300 animals for release over the past 20 years. Data gathered through these captive breeding and release efforts are also used to improve both captive management of the species and future reintroduction efforts (http://www.fws.gov/southwest/es/mexicanwolf/).

Perhaps the most dramatic example of *sorta situ* conservation management in New Zealand is the kakapo. This is an example of "bringing the zoo to the animal." This species is one of the most intensively managed in the world: the entire population lives on two predator-free islands off the coast of southern New Zealand and has been managed there since 1983 (Innes et al. 2010). The population reached a nadir of just 48 in 1993, with only 17 females, but has since reached 123. Nest monitoring, translocation, and artificial feeding, along with the removal of all introduced predators and one native predator, have been instrumental in the success of this program (Clout and Craig 1995; Allen and Lee 2006).

For the whooping crane, the optimal captive flock size, allowing for the retention of greater than 90% of the genetic diversity for more than 100 years, has been determined to be 153. Artificial insemination and selective breeding have been applied to disperse genetics and increase fertility in the captive population, resulting in a managed flock that may have more genetic diversity than the last remaining wild flock of whooping cranes (Jones and Lacy 2009).

Box 40.1 A *Sorta Situ* Success: The Black-footed Ferret

The black-footed ferret (*Mustela nigripes*) is considered to be a conservation success story brought about by captive breeding and reintroduction. Thought to be extinct decades ago, the species was repopulated through a small population discovered in 1985. In this case, regular introduction of animals produced by natural and assisted breeding has proven to be a useful strategy for the maintenance of genetic diversity in this small population (Wisely et al. 2008). Through the recovery of the species, disease has played a major roll. Canine distemper in captive ferrets and sylvatic plague in wild prairie dog colonies and recently in black-footed ferrets have severely set back the program's success at re-establishing the species in the past (Matchett et al. 2006). Intense management of wild populations has included health monitoring, vaccination, surveillance of associated indicator species (coyote, fox, and badger), and identification and monitoring of recaptured adults. Since 1991, 19 specific black-footed ferret reintroduction projects have been conducted across eight U.S. states, Canada, and Mexico. All reintroductions from 1991 to 1996 continue to be occupied by ferrets, and half of all introductions to date are considered "successful" (i.e., self-sustaining with 30 or more breeding adults capable of supporting other sites with translocations) or "improving" (i.e., increasing population) (33% and 17%, respectively). As of 2010, an estimated 1,500 ferrets are living and surviving in the wild across prairie dog habitat with no fewer than 30 reproductive adults in each population. This program has meant an investment of over $US30 million since 1981 and the commitment of many federal, tribal, and state biologists, ecologists, veterinarians, non-governmental organizations, zoos, and private landowners (http://www.defenders.org/programs_and_policy/wildlife_conservation/imperiled_species/black-footed_ferret/background_and_recovery.php).

Health Concerns in Small and Augmented Populations

While the augmentation of wild populations using captive management techniques has to date allowed some species to escape extinction, it underscores the delicate balance of *sorta situ* population management, as it carries its own risks to the survival of the population (Swaisgood et al. 2006). In planning reintroduction of a captive-bred species, managers must consider the implications of captive management techniques on survivability in the wild, and the need to monitor health in the species community once captive-bred animals enter the wild population.

It is well established that the smaller the population, the more susceptible it is to devastation due to disease and other factors. Often, metapopulations—regional populations comprising fragmented subpopulations—persist due to the occurrence of limited dispersal between patches, despite the instability of local subpopulations (Levins 1969). Demographic, environmental, and genetic stochasticity and natural catastrophes have been identified as sources of uncertainty for the determination of a minimum viable population (Shaffer 1981).

When small populations are managed in single-species enclosures in high density, they tend to be susceptible to disease outbreaks much like production animals. Recently, reintroduction efforts for the masked bobwhite quail (*Colinus virginianus ridgwayi*) experienced such a phenomenon (Aguilar et al. 2008; Pacheco et al. 2008). The Buenos Aires Wildlife Refuge is located on the U.S.–Mexico border and was established, in part, to provide habitat for the masked bobwhite quail, populations of which are rapidly declining. In the past 15 years, over 31,000 released masked bobwhite quail were produced at the refuge by a flock of captive quail kept as an assurance colony. In 2007, the flock was reduced by 50% due to a multifaceted disease outbreak in the holding facility, and devastation of the released flock by avian malaria has further complicated the possibility of continuing reintroduction attempts in the region (Andreína Pacheco et al., in press).

Many captive populations are also proving to be unsustainable—fluctuating based on intense competition for limited resources, trends in species interest, and space restrictions—and subject to unique health concerns. Historically, animals have been genetically managed in captivity in small groups, and when their groups exceed available holding space, breeding is prevented. Males are held in isolation, females develop fertility problems associated with failure to carry and deliver offspring on a regular basis, and the fitness of the population declines. In cooperatively breeding species such as African wild dogs (*Lycaon pictus*), the loss of a single dominant breeder can result in prolonged disruptions in breeding. In many species, such as great apes and wild equids, males managed in bachelor groups suffer injury and social stresses not natural for their species. Often, breeding is prevented on a broad scale across zoological institutions, leading to captive population crises. While the genetic and demographic models for sustainable animal populations are viable, the reality of institutional needs and limitations interfere with these models reaching fruition. Of benefit to global populations would be the consideration of the *sorta situ* scale in designing management programs: the provision of more space, larger and more natural groupings, establishment of source and sink populations, and conditions conducive to the development of more adaptive traits would improve health and reproductive potential, as well as better establish populations that may be destined for reintroduction.

Understanding Health and Husbandry Needs

Freshwater mussels of the order Unionoidea are the most imperiled group of animals in North America: over 75% of species are threatened, endangered, of special concern, or extinct (Williams 1993). Yet our lack of understanding of their basic physiological needs has limited our ability to improve their survival. Increasingly, mussel populations are relocated to refuges to protect them from construction zones and invasive mussel colonization, or to recolonize following pollution events and extirpation (Cope and Waller 1995). However, while they are normally long-lived animals, a high proportion of these mussels relocated or brought into captive propagation settings die within the first year of translocation (Cope et al. 2003). Health evaluation of freshwater mussels has historically been limited to behavioral changes and mortality rates. Recently, researchers have begun to develop a systematic approach to the evaluation and monitoring of health and stress in captive freshwater mussels (Wolfe et al. 2008; Burkhard et al. 2009). This diagnostic capability allows better health care, improved understanding of the health concerns of mussels in captivity and in the wild, improved assessment the health of an aquatic habitat through the health of its inhabitants, and improvements in our efforts to conserve these imperiled animals, whose existence is crucial to the health of our freshwater habitats.

Nutritional Challenges

Populations managed *ex situ* on large landscapes are exposed not only to parasites and infectious diseases to which they are naïve, but also to novel forages. In some cases, a particular species may have an unexpected reaction to a plant or toxin—for example, Eld's deer (*Cervus eldi thamin*) exposed to endophyte-infested tall fescue grass, *Festuca arudinacea* (Wolfe et al. 1998). Tall fescue is a hardy, high-yield, cool-season perennial grass and is the most cultivated grass fed to beef cattle in the United States (Alderson and Sharp 1993). The hardiness of fescue grass is further improved by a symbiotic relationship with the endophyte fungus *Neotyphodium* (*Acremonium*) *coenophialum*, which produces a toxic ergot alkaloid. In cattle, while fescue toxicosis most commonly presents as rough hair coat, heat stress, suppressed appetite, poor growth, or reduced calving rates, some animals experience tail tip or hoof sloughing and fat necrosis due to peripheral vasoconstriction. In Eld's deer, abdominal fat necrosis is the most common manifestation and can be so severe as to cause ureteral blockage, uremic crisis, and death. Interestingly, this phenomenon has been noted only in female Eld's deer, presumably due to seasonal rut-related weight loss in males.

Many decades of feeding wild animals in captivity have provided us with a tremendous database of health problems associated with incorrect assumptions about, and deviations from, native diets. As recently as 2005, a review of health problems in captive giraffes (*Giraffa camelopardalis*) has linked the feeding of common ruminant diets to problems such as rumen acidosis, poor body condition, phytobezoars, urolithiasis, hoof problems, and peracute mortality syndrome (Schmidt 2005). In the wild, the giraffe diet comprises primarily *Acacia* and *Combretum* leaves (Pellew 1984), while in captivity this species has typically been offered a diet of hay supplemented with 16% fiber pellets. This low-fiber, high-carbohydrate diet, when fed to giraffe and other browsing species, has been shown to be an inappropriate replacement for browse, resulting in altered volatile fatty acid production in the rumen and ultimately in a host of metabolic problems. Institutions housing browsing ruminants are therefore challenged to provide browse, or a suitable replacement, as a significant proportion of the diet in temperate areas where browse is unavailable for much of the year.

Even when animals live in their native habitat on native food sources, confinement can lead to health problems and population changes when forage is limited and migration is prevented. Rothschild's giraffe (*G. c. rothschildi*) are limited to five viable populations in Kenya and Uganda. In Lake Nakuru National Park, a 50% population decline of Rothschild's giraffe between 1995 (127) and 2002 (62) has been attributed to a dietary change caused by events following the drought resulting from the 1993–95 El Niño Southern Oscillation. Brenneman et al. (2009) suggested that the drought reduced the availability of browse and limited carrying capacity of the park. Due to the limited availability of forage, giraffes were restricted to smaller areas of acacia woodlands and were forced to overgraze the acacia, eating more bark and therefore consuming higher levels of tannin than would normally be tolerated. High levels of concentrated tannins in their diet led to physiologic compromise, particularly in young giraffe, which underwent increased predation by lions in 2001 and 2002. In this case, isolation of a population and disruption of potential migration routes, in combination with climatic events, led to a rapid population decline for what appear to be dietary reasons.

Many species managed *ex situ* are exposed to forages and other feeds in a quantity or cycle that is unnatural. Persian onagers (*Equus hemionus onager*), in their native semi-desert habitat in Iran, forage on relatively poor-quality grasses during the warm season and browse on trees and shrubs when grass is unavailable. Managed on pastures in captivity, this species often encounters year-round lush grasses and/or hay supplemented with pelleted concentrates. In the absence of a "lean" season, these animals have a tendency to develop overabundant fat stores. Interestingly, postmortem incidental findings in obese Persian onagers at The Wilds often include excess liver iron stores as well as indications of liver function compromise (B. Wolfe and E. Blumer unpublished data 2010), which may be associated with obesity. In humans, iron overload has been found to be associated with obesity and insulin resistance (Moirand et al. 1997; Ferranini 2000; Fargion et al. 2005), and in domestic horses, obesity and insulin resistance have been linked to systemic inflammation (Vick et al. 2006; Adams et al. 2009), which can lead to changes in iron metabolism (Borges et al. 2007). The black rhinoceros (*Diceros bicornis*) has also been known to develop iron storage disease in captivity (Smith et al. 1995; Paglia et al. 2001; Dierenfeld et al. 2005; Dennis et al. 2007). The association of obesity and iron overload is currently being investigated in captive and wild black rhinoceroses (P.M. Dennis and M.M. Vick personal communication 2010).

IMPROVING *SORTA SITU* MANAGEMENT FOR SPECIES CONSERVATION

Creating Buffer Zones Around Protected Habitats

Clearly, habitat fragmentation and destruction are having many serious effects on threatened species. Using wildlife management, veterinary care, training, and education, conservationists are working toward mitigating the impacts of fragmentation on species whose survival will necessarily be within small, often isolated, habitat patches. A key example of this is the Atlantic Forest of Brazil, the most endangered rainforest on the planet, with only 2% of its original extent remaining. Within these forest fragments are some of world's most endangered wildlife and plant species, including the black lion tamarin (*Leontopithecus chrysopygus*). A buffer zone is being developed around the protected area in conjunction with an effort to examine the health of individuals, the risk of disease transmission among fragments, and the viability of black lion tamarin metapopulations inhabiting this rainforest. This buffer zone is made up of extremely small fragments of forest, each one too small to sustain a viable population of tamarins. However, this *sorta situ* population has to date avoided extinction due to meticulous management. Ongoing work in Brazil involves the augmentation of the wild population with captive-bred tamarins and research studies aimed at protecting the remaining forest (Valladares-Padua et al. 2002).

Carnivores and other species that range widely have a unique challenge in protected areas. In these reserves, and particularly those with high perimeter-to-area ratios, the majority of large carnivore mortalities are due to human interaction when the animals roam outside the reserve, creating a population sink for these species. For such populations, reserves of greater size, or buffer zones, are preferable for their long-term conservation (Woodroffe and Ginsberg 1998; Baeza and Estades 2010).

Restoring Habitat Quality in Protected Areas

In much of the developing world, human population growth, habitat encroachment, and competition with domestic livestock are having an increasing effect on wildlife populations and habitats. The interactions of these natural and "man-made" processes are often unpredictable. However, experiments in habitat improvement for targeted species and populations are showing how agricultural and forestry practices can be used to restore habitat quality. In Indonesia's Ujung Kulon National Park, the International Rhino Foundation and its partners are working to expand the habitat for the critically endangered Javan rhino (*Rhinoceros sondaicus*). Their plan is to expand rhino habitat in Indonesia by creating a 4,000-hectare research and conservation area adjacent to the currently restricted rhino habitat in Ujung Kulon. This effort will manage the new area to increase/improve rhino "necessities"—water, wallows, saltlicks, and appropriate edible vegetation—to ensure that this area can support an expanded rhino population. This will include replanting natural forest vegetation with rhino food plants in some areas, and carefully implementing controlled slash-and-burn patch management in designated and closed forest areas to promote regeneration of edible plants for rhinos. The project also includes aggressive removal of *Arenga* palm, an invasive species that competes with many key rhino food plants (S. Ellis personal communication 2011).

Using Information Gained from Captive Management for Wild Species Conservation

With the ever-diminishing difference between the ecology of wild and captive populations comes opportunity. As conservation medicine practitioners we have always been charged with using what we learn about wild populations to improve the health and welfare of captive animals. Today, this evolving multi-perspective approach to animal management works in both directions on the spectrum of *sorta situ*. What we learn about captive populations in various management settings can be used to better understand and improve the preservation of wild animals. For instance, a recent study of anesthetic methods in Sichuan takin (*Budorcas taxicolor*) was conducted at The Wilds to prepare for anesthesia of wild takin on a reserve in Sichuan, China, where Chinese and American colleagues are currently studying the behavioral ecology of this little-studied species. In captivity, takin are anesthetized using medetomidine and butorphanol, providing a relatively safe general anesthesia, albeit with slow induction. The large habitats where takin reside at The Wilds allow comparison of anesthetic regimens emphasizing the rate of induction by measuring time and distance traveled from induction to immobilization. This study resulted in an anesthetic regimen that transferred well to the wild takin populations in mountainous Sichuan province, where slow induction and failure to find an animal following the administration of anesthetic could result in mortality.

CONCLUSIONS

The scientific field of conservation medicine is constantly evolving, and with it the methods and assumptions by which we practice. As the discrepancy between "wild" and "captive" animals becomes indistinct, and the management of *in situ* and *ex situ* populations becomes more commonly a spectrum of "*sorta situ*"—neither one nor the other—we must become better able to predict the response of animal populations to new environments in order to protect and conserve them. Our approach to healthcare in populations on the *sorta situ* spectrum has ranged from individual-oriented intensive management in zoological institutions to simple observation and monitoring of wild populations. Today, we must be more strategic and we must have more of an impact. Our efforts to protect populations through translocation or restriction, regardless of the amount of space available, should therefore include an improved ability to predict the effects of geographic barriers and new habitat inputs on animal populations, both under current circumstances and in the case of changing climatic conditions. Conservation efforts must use transdisciplinary approaches to consider entire ecosystems in their management plans, rather than simply single-species populations. Finally, management of animals on large landscapes in our current global reality must take into account the people, agriculture, domestic animals, and disease vectors incumbent on the landscape. To accomplish this monumental task requires more than a hybrid of wild and captive management approaches: it requires a singularity of purpose encompassing a broad spectrum of scientific specialties and social paradigms, which will challenge our breadth and global cooperation into the next millennium.

REFERENCES

Adams, A.A., M.P. Katepalli, K. Kohler, S.E. Reedy, J.P. Stilz, M.M. Vick, B.P. Fitzgerald, L.M. Lawrence, and D.W. Horohov. 2009. Effect of body condition, body weight and adiposity on inflammatory cytokine responses in old horses. Vet Immunol Immunopathol 127:286–294.

Aguilar, R.F., D. Cohan, M. Hunnicut, J.P. Carroll, and M. Garner. 2008. Teetering on the brink: a massive mortality episode in the endangered masked bobwhite quail (*Colinus virginianus ridgwayii*). Proceedings of the American Association of Zoo Veterinarians, pp. 178–179. Los Angeles, California.

Aguirre, A.A. 2009. Biodiversity and human health. EcoHealth 6:153–156.

Aguirre, A.A., and M.C. Pearl. 2004. New technology and *sorta situ*: Conservation medicine linking captive and wildlife populations. Proceedings of the AAZV/AAWV/WDA Joint Annual Conference, pp. 453–455. San Diego, California.

Alderson, J., and W.C. Sharp. 1993. Grass varieties in the United States. Agricultural Handbook No. 170. United States Department of Agriculture, Washington, D.C.

Allen, R.B., and W.G. Lee. 2006. Biological invasions in New Zealand. Ecological Studies Series 186, Springer Verlag, Berlin, Germany.

Arneberg, P. 2001. An ecological law and its macroecological consequences as revealed by studies of relationships between host densities and parasite prevalence. Ecography 24:352–358.

Baeza, A., and Estades, C.F. 2010. Effect of the landscape context on the density and persistence of a predator population in a protected area subject to environmental variability. Biol Conserv 143(1):94–101.

Bode, M., and B. Wintle. 2009. How to build an efficient conservation fence. Cons Biol 24:182–188.

Bolger, D.T., W.D. Newmark, T.A. Morrison, and D.F. Doak. 2008. The need for integrative approaches to understand and conserve migratory ungulates. Ecol Lett 11:63–77.

Borges, A.S., T.J. Divers, T. Stokol, and O.H. Mohammed. 2007. Serum iron and plasma fibrinogen concentrations as indicators of systemic inflammatory diseases in horses. J Vet Internal Med 21:489–494.

Brenneman, R.A., R.K. Bagine, D.M. Brown, R. Ndetei, and E.E. Louis, Jr. 2009. Implications of closed ecosystem conservation management: the decline of Rothschild's giraffe (*Giraffa camelopardalis rothschildi*) in Lake Nakuru National Park, Kenya. Afr J Ecol 47:711–719.

Brooks, T.M., S.L. Pimm, and J.O. Oyugi. 1999. Time lag between deforestation and bird extinction in tropical forest fragments. Conserv Biol 13:1140–1150.

Burkhard, M.J., S. Leavell, R.B. Weiss, K. Kuehnl, H. Valentine, G.T. Watters, and B.A. Wolfe. 2009. Analysis and cytologic characterization of hemocytes of freshwater mussels (*Quadrula* sp.). Vet Clin Path 38:426–436.

Catt, D.C., D. Dugan, R.E. Green, R. Moncrieff, R. Moss, N. Picozzi, R.W. Summers, and G.A. Tyler. 1994. Collisions against fences by woodland grouse in Scotland. Forestry 67:105–118.

Chape, S., J. Harrison, M. Spaulding, and I. Lysenko. 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global diversity targets. Phil Trans Royal Soc B 360:443–455.

Chase, M.J., and C.R. Griffin. 2009. Elephants caught in the middle: impacts of war, fences and people on elephant distribution and abundance in the Caprivi Strip, Namibia. Afr J Ecol 47:223–233.

Chomel, B.B., A. Belotto, and F.-X. Meslin. 2005. Wildlife, exotic pets and emerging zoonosis. Emerg Infect Dis 13:6–11.

Clout, M.N., and J.L. Craig. 1995. The conservation of critically endangered flightless birds in New Zealand. Ibis 137:S181–S190.

Clubb, R., and G. Mason. 2003. Captivity effects on wide-ranging carnivores. Nature 435:473–474.

Cohn, J.P. 2007. The environmental impacts of a border fence. Bioscience 57:doi:10.1641/B570116.

Conde, D.A., N. Flesness, F. Colchero, O.R. Jones, and A. Scheuerlein. 2011. An emerging role of zoos to conserve biodiversity. Science 331:1390–1391.

Cope, W.G., and D.L. Waller. 1995. Evaluation of freshwater mussel relocation as a conservation and management strategy. Reg Rivers-Res Manag 11:147–155.

Cope, W.G., M.C. Hove, D.L. Waller, D.J. Hornback, M.R. Bartsch, L.A. Cunningham, H.L. Dunn, and A.R. Kapuscinski. 2003. Evaluation of relocation of unionid mussels to in situ refugia. J Molluscan Stud 69:27–34.

Cranfield, M., L. Gaffikin, J. Sleeman, and M. Rooney. 2002. The mountain gorilla and conservation medicine. *In* Aguirre, A.A., R.S. Ostfeld, G.M. Tabor, C. House, and M.C. Pearl, eds. Conservation medicine: ecological health in practice, pp. 282–296. Oxford University Press, New York.

Creel, S., and D. Christianson. 2008. Relationships between direct predation and risk effects. Trends Ecol Evol 23:194–201.

Creel, S., and D. Christianson. 2009. Wolf presence and increased willow consumption by Yellowstone elk: implications for trophic cascades. Ecology 90:2454–2466.

Cross, P.C., E.K. Cole, A.P. Dobson, W.H. Edwards, K.L. Hamlin, G. Luikart, A.D. Middleton, B.M. Scurlock, and P.J. White. 2010. Probable causes of increasing brucellosis in free-ranging elk of the Greater Yellowstone Ecosystem. Ecol Appl 20:278–288.

Cumming, D., H. Biggs, M. Kock, N. Shongwe, S. Osofsky and members of the AHEAD-Great Limpopo TFCA Working Group. 2007. The AHEAD (Animal Health for Environment and Development)-Great Limpopo Transfrontier Conservation Area (GLTFCA) Programme: Key questions and conceptual framework revisited. Wildlife Conservation Society, The Bronx, New York. http://wcs-ahead.org/workinggrps_limpopo.html

Daszak, P., and A.A. Cunningham. 2002. Emerging infectious diseases: a key role for conservation medicine. *In* A.A. Aguirre, R. Ostfeld, G.M. Tabor, C. House, and M.C. Pearl, eds. Conservation medicine: ecological health in practice, pp. 40–61. Oxford University Press, New York.

Daszak, P., A.A. Cunningham, and A.D. Hyatt. 2000. Emerging infectious diseases of wildlife –threats to biodiversity and human health. Science 287: 443–449.

Demarais, S., J.T. Baccus, and M.S. Traweek, Jr. 1998. Nonindigenous ungulates in Texas: long-term population trends and possible competitive mechanisms. Trans North Am Wildl Nat Res Conf 63:49–55.

Dennis, P.M., J.A. Funk, P.J. Rajala-Schultz, E.S Blumer, R.E. Miller, T.E. Wittum, and W.J.A. Saville. 2007. A review of some of the health issues of captive black rhinoceroses (*Diceros bicornis*). J Zoo Wildl Med 38:509–517.

Dierenfeld, E.S., S. Atkinson, A.M. Craig, K.C. Walker, W.J. Streich, and M. Clauss. 2005. Mineral concentrations in serum/plasma and liver tissue of captive and free-ranging rhinoceros species. Zoo Biol 24:51–72.

Dobson, A. 2009. Food web structure and ecosystem services: insights from the Serengeti. Phil Trans Royal Soc B 364:1665–1682.

Fargion, S., P. Dongiovanni, A. Guzzo, S. Colombo, L. Valenti, and A.L. Fracazani. 2005. Iron and insulin resistance. Aliment Pharm Therapeut 22:61–63.

Ferranini, E. 2000. Insulin resistance, iron and the liver. Lancet 355:2181–2182.

Ferraz, G., G.J. Russell, P.C. Stouffer, R.O. Bierregaard, Jr., S.L. Pimm, and T.E. Lovejoy. 2003. Rates of species loss from Amazonian forest fragments. Proc Natl Acad Sci 100:14069–14074.

Flesch, A.D., C.W. Epps, J.W. Cain III, M. Clark, P.R. Krausman, and J.R. Morgart. 2010. Potential effects of the United States-Mexico border fence on wildlife. Conserv Biol 24:171–181.

Fortin, D., H.L. Beyer, M.S. Boyce, D.W. Smith, T. Duchesne, and J.S. Mao. 2005. Wolves influence elk movements: behavior shapes a trophic cascade in Yellowstone National Park. Ecology 86: 1320–1330.

Heithaus, M.R., A.J. Wirsing, D. Burkholder, J. Thompson, and L.M. Dill. 2009. Towards a predictive framework for predator risk effects: the interaction of landscape features and prey escape tactics. J Anim Ecol 78:556–562.

Hoole, A., and F. Berkes. 2010. Breaking down fences: recoupling social–ecological systems for biodiversity conservation in Namibia. Geoforum 41:304–317.

Hudson, R.J., K.R. Drew, and L.M. Baskin. 1989. Wildlife production systems: economic utilisation of wild ungulates, Cambridge University Press, United Kingdom.

Huerta-Patricio, E., K.D. Cameron, G.N. Cameron, and R.A. Medellin. 2005. Conservation implications of exotic game ranching in the Texas Hill Country. *In* V. Sanchez-Cordero and R.A. Medellín. eds. Contribuciones mastozoologicas en homenaje a Bernardo Villa. Ch. 21, pp. 237–252. Instituto de Biología and Instituto de Ecología, UNAM, CONABIO, México.

Innes, J., D. Kelly, J.M. Overton, and C. Gillies. 2010. Predation and other factors currently limiting New Zealand forest birds. New Zeal J Ecol 34:86–114.

IUCN. 1994. Guidelines for protected area management categories. International Union for the Conservation of Nature. Gland, Switzerland.

Kalema-Zikusoka, G.K., J.M. Rothman, and M.T. Fox. 2005. Intestinal parasites and bacteria of mountain gorillas (*Gorilla beringei beringei*) in Bwindi Impenetrable National Park, Uganda. Primates 46:59–63.

Kellert, S.R., and E.O. Wilson. 1993. The biophilia hypothesis. Cambridge Island Press/Shearwater, Washington, D.C.

Lees, C.M., and J. Wilcken. 2009. Sustaining the ark: the challenges faced by zoos in maintaining viable populations. Int Zoo Yearbook 43:6–18.

Levins, R. 1969. Some demographic and genetic consequences of environmental heterogeneity for biological control. Bull Entomol Soc Am 15:237–240.

Li, C.W., Z.G. Jiang, S.H. Tang, and Y. Zeng. 2007. Influence of enclosure size and animal density on fecal cortisol concentration and aggression in Pere David's deer stags. Gen Comp Endocrinol 151:202–209.

Jones, K., and B. Lacy. 2009. Whooping crane master plan. International Whooping Crane Recovery Team, Baraboo, Wisconsin.

Joppa, L.N., S.R. Loarie, and S.L. Pimm. 2008. On the protection of "protected areas." Proc Natl Acad Sci USA 105:6673–6678.

Matchett, M.R., D.E. Biggins , V. Carlson, B. Powell, and T. Rocke. 2010. Enzootic plague reduces black-footed ferret (*Mustela nigripes*) survival in Montana. Vector-borne Zoonot 10:27–35.

Metrione, L., and J.C. Harder. 2009. The effects of environment on the social development and reproductive success of adolescent and adult female southern white rhinoceros. Sixth Rhino Keeper Workshop, Rhino Keeper Association, Bush Gardens, Tampa, Florida.

Moirand, R., A.M. Mortaji, O. Loréal, F. Paillard, P. Brissot, and Y. Deugnier. 1997. A new syndrome of liver iron overload with normal transferrin saturation. Lancet 349:95–97.

Mungall, E., and W.J. Sheffield. 1994. Exotics on the range. Texas A&M University Press, College Station, Texas.

Nagy, D.W. 2005. Parelaphostrongylosis and other parasitic diseases of the ruminant nervous system. Vet Clin North Am 20:393–412.

Newmark, W.D. 1996. Insularization of Tanzanian parks and the local extinction of large mammals. Conserv Biol 10:1549–1556.

Newmark, W.D. 2008. Isolation of African protected areas. Frontiers Ecol Environ 6:321–328.

Osofsky, S.A., D.H. Cumming, and M.D. Kock. 2008. Transboundary management of natural resources and the importance of a "one health" approach: perspectives on southern Africa. *In* E. Fearn and K.H. Redford, eds. State of the wild 2008–2009: a global portrait of wildlife, wildlands, and oceans. Island Press, Washington, D.C. http://www.wcs-ahead.org/print.html

Pacheco, M.A., A.A. Escalante, M.M. Garner, G.A. Bradley, and R.F. Aguilar. 2011. Haemosporidian infection in captive masked bobwhite quail (*Colinus virginianus ridgwayi*), an endangered subspecies of the northern bobwhite quail. Vet Parasitol 182:113–120.

Pacheco, M.A., A. Escalante, K. Orr, M. Garner, and R.F. Aguilar. 2008. Acute mortality and malaria in captive masked bobwhite quail. Proc Wildl Soc New Zeal Vet Assoc Conf., p. 11. Palmerston North, New Zealand.

Paglia, D.E., D.E. Kenny, E.S. Dierenfeld, and I.H. Tsu. 2001. Role of excessive maternal iron in the pathogenesis of congenital leukoencephalomalacia in captive black rhinoceroses (*Diceros bicornis*). Am J Vet Res 62:343–349.

Pellew, R.A. 1984. The feeding ecology of a selective browser, the giraffe (*Giraffa camelopardalis tippelskirchi*). J Zool 202:57–81.

Phillips, A. 2003. Turning ideas on their head: the new paradigm for protected areas. Paper written for Fifth World Parks Congress. www.uvm.edu/conservationlectures/vermont.pdf. Last accessed April 18, 2011.

Sayre, N.F., and R.L. Knight. 2009. Potential effects of United-States-Mexico border hardening on ecological and human communities in the Malpai borderlands. Conserv Biol 24:345–348.

Schmidt, D. (ed.). 2005. Giraffe Nutrition Workshop Proceedings, PMI Nutrition International, Lincoln Park Zoo, Chicago, Illinois.

Schmitt, S.M., S.D. Fitzgerald, T.M. Cooley, C.S. Bruning-Fann, L. Sullivan, D. Berry, T. Carlson, R.B. Minnis, J.B. Payeur, and J. Sikarskie. 1997. Bovine tuberculosis in free-ranging white-tailed deer from Michigan. J Wildl Dis 33:749–758.

Shaffer, M.L. 1981. Minimum population sizes for species conservation. BioScience 31:131–134.

Sih, A., D.I. Bolnick, B. Luttbeg, J.L. Orrock, S.D. Peacor, L.M. Pintor, E. Preisser, J.S. Rehage, and J.R. Vonesh. 2010. Predator-prey naivete, antipredator behavior, and the ecology of predator invasions. Oikos 119:610–621.

Smith, J.E., P.S. Chavey, and R.E. Miller. 1995. Iron metabolism in captive black (*Diceros bicornis*) and white (*Ceratotherium simum*) rhinoceroses. J Zoo Wildl Med 26:525–531.

Soulé, M., M. Gilpin, W. Conwa, and T. Foose. 1986. The millennium ark: how long a voyage, how many staterooms, how many passengers? Zoo Biol 5:101–113.

Swaisgood, R.R., D.M. Dickman, and A.M. White. 2006. A captive population in crisis: Testing hypotheses for reproductive failure in captive-born southern white rhinoceros females. Biol Conserv 129:468–476.

Teer, J.G., L.A. Renecker, and R.J. Hudson. 1993. Overview of wildlife farming and ranching in North America. Trans North Am Wildl Nat Res Conf 58:448–459.

Valladares-Padua, C., S.M. Padua, and L. Cullen Jr. 2002. Within and surrounding the Morro do Diablo State Park: biological value, conflicts, mitigation and sustainable development alternatives. Environ Sci Policy 5:69–78.

Vicente, J., U. Höfle, I. García Fernández-De-Mera, and C. Gortazar. 2007. The importance of parasite life history and host density in predicting the impact of infections in red deer. Oecologia 152:155–164.

Vick, M.M., B.A. Murphy, D.R. Session, S.E. Reedy, E.L. Kennedy, D.W. Horohov, R.F. Cook, and B.P. Fitzgerald. 2006. Effects of systemic inflammation on insulin sensitivity in horses and inflammatory cytokine expression in adipose tissue. Am J Vet Res 69:130–139.

Western, D. 2002. In the dust of Kilmanjaro. Island Press, Washington.

Wildlife Society. 2010. Final position statement: the impact of border security measures on wildlife, 2 pp. TWS, Washington, D.C. http://joomla.wildlife.org/index.php?option=com_content&task=view&id=117&Itemid=299. Last accessed April 18, 2011.

Williams, E.S., and M.W. Miller. 2002. Chronic wasting disease in deer and elk in North America. Rev Sci Tech Off Int Epiz 21:305–316.

Williams, J.D., M.L. Warren, K.S. Cummings, J.L. Harris, and R.J. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18:6–22.

Wilson, P.R. 2002. Advances in health and welfare of farmed deer in New Zealand. New Zeal Vet J 50:S105–S109.

Wilson, K.J. 2004. Flight of the Huia: ecology and conservation of New Zealand's frogs, reptiles, birds and mammals. Canterbury University Press, Christchurch, New Zealand.

Wisely, S.M., R.M. Santymire, T.M. Livieri, S. Mueting, and J.G. Howard. 2008. Genotypic and phenotypic consequences of reintroduction history in the black-footed ferret (*Mustela nigripes*). Conserv Gen 9:389–399.

Wolfe, B.A., M. Bush, S.L. Monfort, S.L. Mumford, A. Pessier, and R.J. Montali. 1998. Abdominal lipomatosis attributed to tall fescue toxicosis in deer. J Am Vet Med Assoc 213:1783–1785.

Wolfe, B.A., M.J. Burkhard, S. Leavell, R.B. Weiss, K. Kuehnl, H. Valentine, G.T. Watters, and B. Bergstrom. 2008. The clam exam: clinical pathology of freshwater mussels, p. 186. Proc Am Assoc Zoo Vet, Los Angeles, California.

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