MITIGATING THE IMPACTS OF HUMAN LAND-USE CHANGE ON BIODIVERSITY: WITH A FOCUS ON LARGE MIGRATORY HERBIVORES

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This dissertation is approved, and it is acceptable in quality and form for publication:

Approved by the Dissertation Committee:

Scott Collins, Chairperson

Marcy Litvak

Blair Wolf

Kevin Kirkman
MITIGATING THE IMPACTS OF HUMAN LAND-USE CHANGE ON
Biodiversity: With a Focus on Large Migratory Herbivores

By

Kina Rebekah Murphy

B.S., Conservation Science, College of Santa Fe, 2000
M.C.R.P., Natural Resource Management, University of New Mexico, 2006

Dissertation

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Biology

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Dedication

This is dedicated to my daughters, Keda and Zakhiah Kanye, who have supported me and put up with me every step of the way. You have given me the strength and courage to be so much more than I imagined.

Thank you for being my light.

&

To my grandmother Eurith Harper who died at 95, just months before I completed this.
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MITIGATING THE IMPACTS OF HUMAN LAND-USE CHANGE ON BIODIVERSITY: WITH A FOCUS ON LARGE MIGRATORY HERBIVORES

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KINA REBEKAH MURPHY
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Ph.D., Biology, University of New Mexico, 2016

ABSTRACT

Land-use change, commercial over-harvesting of species, and climate change are recognized as the main drivers of biodiversity loss. As a result, it is estimated that 30% of the planet’s biodiversity may go extinct by 2050. This dissertation focuses on how to mitigate the impacts of land-use change on biodiversity. I focus on large migratory herbivores because they are among the most heavily impacted by global change due to their large home range requirements. Habitat fragmentation, illegal hunting, and human-wildlife conflicts are among the biggest threats to large herbivores and result from land-use change. For this reason, my first chapter focuses on monitoring the daily and hourly movement patterns of large herbivores to and from water resources to determine if humans can modify their behaviors in ways that will reduce conflict and habitat fragmentation. The data suggest that herbivore movement patterns can be predicted, and that humans can delineate wildlife movement corridors and design development projects that minimize impacts on large herbivores. The conversion of land to human use has been shown to increase illegal hunting. However, I discuss how a hunting ban has led to loss of local livelihoods magnifying the need for illegal
hunting, compelled people to obtain more livestock to increase their incomes, and displaced rural people leading to increased land-use change. I explore the other potential drivers of species loss and illustrate how mitigating these drivers and valuing wildlife resources in a way that supports rural communities can have a more positive impact on biodiversity. I then explore whether it is possible for large mining projects to result in no-net loss or a net gain in biodiversity. I use the Oyu Tolgoi mine in Mongolia as a case study of how mitigation and offset programs can be used to increase biodiversity. I then suggest that focusing mitigation and offset programs on restoring and enhancing ecological processes and whole landscapes may be more effective than a focus solely on threatened and endangered species.
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Chapter 1: Introduction

It is estimated that by 2050, over 30 percent of the planet’s biodiversity may go extinct (WWF 2014; Vitousek et al. 1997). Land-use change, commercial over-harvesting of species, and climate change are recognized as the main drivers of biodiversity loss (UNEP 2010; IPCC 2014). The sustainable use of resources has been identified as the solution (UNEP 2010) and has been promoted worldwide for half a century, yet our vast impacts on the biomes of the world have not been significantly mitigated. Since the industrial revolution humans have become the main driver of global change by pushing the earth toward abrupt critical transitions that would otherwise occur over tens of thousands of years (Steffen et al. 2007). Some of these transitions may be irreversible and could create environmental conditions that are not conducive to human development (IPCC 2014).

It is suggested that we are approaching global boundaries for land use change (Rockstrom et al. 2009). Currently, 43% of the earth’s terrestrial surface has been converted to human-use. This includes conversion of land to agriculture, ranching, cities and roads. According to the WWF (2014) ‘Living Planet Index’, land-use change and habitat loss associated with other anthropogenic drivers have been shown to be responsible for 44% of global biodiversity loss, followed by exploitation, which is responsible for 37% of global biodiversity loss. It is hypothesized that when 50% of the earth has been converted to human-use a state change will occur (Barnosky et al. 2012). This number could be as high as 90%, but researchers assume that 50% is a safe estimate given that the last global landscape transition that pushed earth into a biological state change (from the Pleistocene to the Holocene) occurred when only 30% of the Earth’s surface went from being covered in glacial ice to being ice free (Scheffer et al. 2009; Barnosky et al. 2012).
The melting of glacial ice may appear more significant than human induced land conversion. However, recent studies (Suding & Hobbs 2009) have indicated that land-use change can have the most significant effect on threshold triggers through complex feedback loops that cannot be easily predicted. This is also due to the fact that land degradation creates internally reinforced states (Suding, Gross, & Houseman 2004) that push systems toward undesired trajectories, such as desertification and loss of biodiversity, which impairs ecosystem resilience (Ives & Carpenter 2007). Additionally, humans have effected incremental land-use changes linked to multiple biotic and abiotic processes over periods of time not visible within one or two generations. These changes have, in some cases, caused ecosystems to cross thresholds that have led to alternative states before humans were aware of their impacts (Rockstrom et al. 2009; Lavergne et al. 2010).

Migratory species are among the most heavily impacted by land-use change (Bolger et al. 2008). Due to their large home range requirements, they are among the most endangered taxa on the planet (CMS 2012; CMS 2010). When graphed according to family, home range generally increases with body size, making large migratory species even more vulnerable to extinction than smaller species (Lindstedt et al. 1986; Estes et al. 2011; Peters 1983). This is because large animals have smaller populations than smaller species, and extinction rates increase with decreasing population size (Calder 1984; Peters 1983). Some large herbivores also require access to vast intact ecological processes that span multiple countries, such as rivers and wetlands that filter water and deliver nutrients to riparian landscapes.

The majority of these ecological processes (such as: nutrient cycling and natural succession) are, like large herbivores, threatened by human land-use change and degradation
(IUCN 2007; IPCC 2014; Vanderpost 2006), and are shifting as a result of climate change. Where habitat for large migratory species still exists, migrators are increasingly unable to access critical seasonal habitats due to habitat fragmentation caused by humans (Hopcraft et al. 2014). This leads to human-wildlife conflict (HWC), which further threatens wildlife populations. Similarly, land-use change increases poaching and the exploitation of animals, especially large herbivores. Increased land-use change through grazing also degrades rangeland critical to the survival of large herbivores and increases competition for resources.

My dissertation therefore focuses on: 1) how understanding the local habitat needs of large herbivores can help mitigate habitat fragmentation and human-wildlife conflict caused by land-use change; 2) How restrictive policies, such as hunting bans can have the unintended consequence of exacerbating biodiversity loss by increasing illegal hunting and land-use change; and 3) how rangeland degradation can lead to the loss of groundwater tables that support multiple trophic interactions. I focus on large migratory species because they are the most impacted by land-use change and exploitation and because the risk of extinction is greatest for large migratory species. They also play a critical role as ecosystem engineers that many other species depend on.

By driving ecological succession, large herbivores transforming woodlands to shrub and grasslands. They increase the productivity and nutrient quality of grasslands by stimulating plant growth at low and moderate grazing intensity (McNaughton 1979; McNaughton et al. 1997, Augustine & McNaughton 1998), and drive grassland ecosystem dynamics by increasing the diversity of mesic grasses (Collins 1987, Hartnett et al. 1996). They also increase resource heterogeneity, and alter community structure through
disturbance and nutrient deposition (McNaughton et al. 1997). As a result, large herbivores have been shown to increase biodiversity and the resilience of whole ecosystems.

Chapter One of my dissertation focuses on how to mitigate the effects of HWC and habitat fragmentation by predicting the movement patterns of large herbivores along the Chobe-Linyanti Riverfront and Wetlands in Botswana. I examine how a better understanding of the movement and resource needs of species can inform development decisions such that negative impacts to species can be avoided. Mitigating human impacts to biodiversity often only requires careful consideration of how losses can be avoided.

Chapter Two explores how the implementation of a hunting ban in Botswana has led to unintended impacts by stimulating additional land-use change, which increases biodiversity loss, and removing rural livelihood mechanisms, which pushes communities toward illegal hunting. I suggest that mitigating the other drivers of species loss and developing methods for valuing Botswana’s wildlife that support rural communities will have a more positive impact on biodiversity than prohibition of hunting.

Chapter Three is a critique of the Oyu Tolgo (OT) Core Biodiversity Monitoring Initiative, that I developed and managed for the Wildlife Conservation Society, whose goal is to increase biodiversity in the area impacted by the OT mine. The mine hopes to increase biodiversity by improving rangeland quality, reducing hunting, and mitigating the mine’s negative impacts on priority biodiversity features. In this chapter I suggest that disturbances to abiotic factors, such as water and soil, should be included in the core biodiversity monitoring because they are critical to the survival of all species and to rangeland health. I suggest that erosion and the subsequent loss of shallow ground water tables are threshold
triggers that can be used to identify broad-scale disturbances and climate drivers that affect ecosystem resilience (Suding & Hobbs 2009) in all human impacted landscapes.

This combined research is an in-depth study of how, through careful consideration of our impacts on species and ecosystems, humans can begin to reduce biodiversity loss and reverse land degradation. It provides an intricate view the movement and assemblage patterns of large herbivores along the Chobe-Linyanti Wetland and suggests methods for mitigating human-wildlife conflict and habitat fragmentation. It uses a hunting ban in Botswana as a case study for how loss of local livelihoods can lead to increased legal hunting and land-use change. Lastly, we explore how to effectively monitor landscapes for biodiversity loss and suggest that monitoring the degradation of abiotic processes may be most effective.
References


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Chapter 2: Can Predicting the Movement and Assemblage Patterns of Large Herbivores Mitigate Human Wildlife Conflict?

Abstract

Large herbivores in southern Africa are declining due to habitat fragmentation associated with land-use change and lack of movement corridors. In response, wildlife frequent human use areas, leading to human-wildlife conflict that exacerbates wildlife declines. While human-wildlife conflict has been associated with the seasonal movement needs of these species, conflicts can be mitigated if humans understand the daily movement needs of large herbivores. We monitored the daily and hourly movement patterns of herbivores along Botswana’s Chobe-Linyanti Wetland between August 2004 and August 2005. We show emerging, predictable patterns in the movement and assemblage patterns of large herbivores based on time of day, season, river morphology, and proximity to other species. Our data suggest that many species access the river at specific times of day; that those times vary depending on season; that river morphology is correlated with richness and abundance of herbivores; and that some species occupy specific niches in time and space to avoid competition for access to water. Our data underscore the value of predicting when and where human-wildlife conflict is likely to occur in order to help conservation professionals delineate appropriate wildlife movement corridors and planners to design development projects that do not impact herbivore movements.

Introduction

Human-wildlife conflict (HWC) is one the main threats to large herbivores in Africa (Elliot et al. 2008; Barns 1996; Ogada et al. 2003). Conflicts with animals not only lead to the killing and relocating of problem animals as humans attempt to protect their lives and
livelihoods, but it also undermines conservation efforts as human sensitivity toward wildlife declines (Elliot et al. 2008; CARACAL 2016; WWF 2016). These conflicts are often a direct result of human caused habitat fragmentation and land-use change (Selier 2015; Ogada et al. 2003). A study conducted by the World Wildlife Fund (Elliot et al. 2008) on multiple continents has shown that HWC can be drastically reduced when species are provided with adequate habitat and where human activities within buffer zones do not attract wildlife. Multiple case studies have shown that buffer zones that do not include farming are a deterrent to wildlife (Elliot et al. & Reed 2008; Selier 2015; Hopcraft et al. 2014).

The frequency of HWC incidents in Africa has fluctuated since the imposition of colonialism in Africa. European and Middle Eastern immigrants implemented aggressive elephant culling programs that drastically reduced HWC and elephant populations (Hoare 1999; Kangwana 1995). The end of the ivory trade and colonialism allowed for the restoration of elephant populations, but human settlements and farming had already increased by this time (Hoare 1999; Kangwana 1995). While human-elephant conflict was not correlated with increases in elephant population, it was correlated with the proximity of humans to elephant habitat (Hoare 1999; Kangwana 1995; Elliot et al. 2008), which increased during colonialism and continues to the present day in parallel with human population increases.

Prior to colonialism in Africa, farming was probably only successful in well-defended villages (Hoare 1999; Laws, Parker & Johnstone 1975). Generations of traditional ecological knowledge (TEK) has shown that humans can avoid HWC by understanding when and where animals are likely to move within daily and seasonal time intervals (Berkes et al., 2000, Turnhout et al., 2012). This can be seen today in Bushmen villages where men wake early in
the morning to assess the location of problem wildlife and report locations to women as they venture out to gather food and thatching materials. Similarly, communities have historically defended crops at night when elephants are likely to raid them. However, behaviors and locations of humans have changed, and as men travel to neighboring villages for work, young people move to urban centers and immigration increases, traditional mechanisms for dealing with HWC are lost (Turvey et al. 2010). Women are left to watch crops, livestock, children, elders and sick individuals and often cannot defend crops (Alexander & Ramotadima 2004). This leads to health and safety issues that exacerbate poverty in these communities (UN Gender Equality Portal 2016).

This pattern can be observed clearly in Northern Botswana, which is home to 128,000 elephants (Chase 2011), and where large herbivores tend to congregate in Botswana’s Chobe National Park as they flee land-use change and hunting in adjacent countries. Nevertheless, habitat within Chobe National Park is not adequate for Botswana’s large population of elephants, resulting in 80 percent of elephant habitat still occurring outside of the park (Chase 2011). Within Botswana, large herbivores are constrained by the proximity of high quality and quantity forage to water, especially during the dry season (Redfern et al. 2003, Fynn & Bonyongo 2011). This causes them to congregate along the Chobe Riverfront and utilize adjacent community areas along the Chobe Wetlands where humans are farming (Omphile 2002).

Rural farmers are suffering tremendous losses to large herbivores and their predators (Botswana Ministry of Local Government 2009, Botswana Ministry of Finance and Development Planning 2003). As a result, the Botswana Department of Wildlife and National Parks (DWNP) implemented a program that relocates and sometimes kills problem
individuals. However, because herbivores are constrained by the proximity of their seasonal forage to water, killing and relocating animals is largely ineffective. We suggest that the daily movement needs of these species may be predictable, which can allow humans to reduce HWC if local residents fully understand wildlife movement requirements, make critical movement corridors available to them, and modify human movements during specific times of the year and at specific times of day.

While extensive research has been done on the wet and dry season habitat needs and migration routes of large herbivores, little knowledge exists on their daily and hourly movement needs and how these change seasonally. To understand these rhythms of resource use, we monitored the daily movement and assemblage patterns of 15 large herbivores (see Table 1) between August 2004 and August 2005, to predict how large herbivores move within the wetland corridor and to explore the possibility of human-wildlife compatibility.

We hypothesize that: 1) animals prefer areas along the riverfront with the highest degree of channel heterogeneity (such as one or several meandering channels, braided channels and alluvial fans existing in the same general location); 2) that some species will prefer deep flowing channels and areas where access is flat rather than sloped; 3) that species richness and abundance will be lower in areas utilized by human communities and where animals are hunted (Selier et al. 2015); 4) that animals will access the river at specific times of day; 5) that those times will vary according to season; and 6) that each species will occupy a specific niche in time and space. We suggest that knowledge of these patterns can be used to modify human behavior in ways that will mitigate HWC and reduce habitat fragmentation for large herbivores.
Materials and Methods

Site. The Chobe-Linyanti wetland is fed by the Kwando River, originating in the Angolan highlands; the Chobe River, which originates from Lake Liambezi in Botswana; and the Zambezi River, which originates in Zambia to the north (Pricope 2012). Inflows from the Kwando and Zambezi Rivers begin flooding the wetland between March and May and range between 6000 million m³ to 9000 million m³ (Pricope 2012). The water is not visible in the wetland until April and continues flooding it throughout Botswana’s dry season, which lasts until September. Floods reach their peak between June and August, and seasonal rains, which begin in October, increase flooding.

The Chobe River acts as a border between Botswana and Namibia. The entire stretch of the Chobe-Linyanti wetland is about 250 km, from Linyanti, at S 17°50’ and E 23°25, to Kazungula, at S 18°28’ and E 25°16’, where Botswana meets Zambia, Zimbabwe, and Namibia. The average temperature in the region ranges between 23°-25°C during summer months and 5°-7°C in winter months. Highest temperatures typically occur in January. The entire stretch of the study area consists of similar habitat types. Mesic grasslands exist in some portions of the flood plain, while sedges and rushes dominate in swampy areas. In upland woodlands Burkea africana, Colophospermum mopane, Combretum spp, Terminalia spp and scattered Adansonia kilima dominate the landscape.

For the purpose of this study, we divided the Chobe Riverfront and wetlands into four sections based on management regimes, some of which have changed due to Botswana’s recent ban on hunting. These sections include: Chobe National Park (CNP) Riverfront, The Chobe Enclave (a community area), Linyanti CNP, and Linyanti Rann (a former community hunting area). The Riverfront and its associated wetlands stretch approximately 141 km, with
Chobe National Park Riverfront stretching 51 km, Linyanti and Rann 50 km, and the Chobe Enclave 40 km.

The Chobe Enclave is a community area sandwiched by Chobe National Park on its east, west and south sides and by the Chobe wetland and Namibia to the north. It consists of 5 main villages, three on the banks of the floodplain and two within the floodplain. Humans in this region are “malapa” (wetland) farmers and rely on seasonal rains and floods to grow their crops. Hunting was not allowed in the Chobe Enclave. Rann, the community hunting area, consisted of one lodge and several remote hunting camps and was only operational during Botswana’s, now banned, hunting season. The Chobe Riverfront is within Chobe National Park and has a moderate number of tourism vehicles that frequent the Riverfront, but animals did not appear deterred by non-threatening vehicles. Similarly, the Riverfront is adjacent to the village of Kasane and animals were not shy of people in this village. Hunting was also not permitted. Linyanti is also within Chobe National Park yet animals were very skittish of vehicles, most likely due to its close proximity to hunting areas (Selier et al. 2015).

Most HWC incidents occur within the Chobe Enclave whose population was 7,000 at the time of this study (2004-2005), and the village of Kasane, whose population was 7,500 at the time of this study. In Kasane, conflict incidents occur between the business center, which is the most densely populated part of the village and is situated on the banks of the Chobe River, and the plateau, where most residents are located. There is a large tract of wooded open space between the business center and the residential area (approximately 1 kilometer). Villagers walk several designated paths within this open space when traveling between home and work or shopping areas. Most people who live in this village are not farmers and have 9-5 jobs. In the Enclave several villages are located just above the floodplain, where the river
channel is shallow and branches off into multiple small channels. In this area, most resident are malapa farmers. Farms are placed on the edge of the wetland in order to take advantage of seasonal floods. Humans move to and from the wetland to tend their crops. The majority of human-wildlife conflicts occur between villages and farms.

The majority of conflict incidents occur as follows: elephants trample and throw people, raid crops, raid water storage facilities, can cause 100% loss of crops and damage other resources. Buffalo trample and impale people and raid crops (Alexander 2004). Medium sized herbivores also raid crops, but this does not occur often.

**Distance sampling.** We monitored the movement and assemblage patterns of large herbivores using Distance Sampling methods (Buckland 2009) between August 2004 and August 2005. Transects were placed parallel to the river channel no more than 500 meters from the channel in each section, and between two and four kilometers apart parallel to the channel. Each study section consists of two parallel transects that span the entire length of the section. Perpendicular transects were placed approximately 2-4.5 kilometers apart and run exactly eight kilometers from the river or wetland into the upland forests. At least two perpendicular transects were driven in each research section, but as many as four were driven in some sections. All transects were driven two times a day: morning 6:00 am-1:00 pm, and afternoon 1:00-8:00 pm on different days. This was done three times (3 mornings and 3 afternoons = 6 different days) in each of the following seasons: The hot-rainy season (December-February), the cold-rainy season (March-May), the cold-dry season (June-August), and the cold-rainy season (September-November).

Each species and the number of individuals observed while driving between one and ten kilometers per hour, was recorded using a GPS, along with distance of each individual
from the road/transect. Distance and the approximate angle from the road to the animal were recorded using a laser rangefinder. Each individual was then recorded as a GPS point and entered into a geographic information system (ArcGis 10.3). The time of day; distance from the river; distance from human settlements; nearest habitat association; species; species number; and aspect (which side of road the animals were on) were recorded while driving each transect. Cloudy and rainy days were also recorded at the beginning of each transect to determine if temperature was affecting animal movements to and from water sources.

Each transect was traversed by vehicle three times in the dry season and three times in the rainy season. However, extremely heavy flooding made the community area inaccessible for at least two months during the rainy season and two months during the dry season when inflows magnified flooding from rain. Transects could not be monitored during these heavy flood stages. The final data analysis account for these missing data points by reducing the number of transects used in data analyses.

**Nearest neighbor.** Nearest neighbor calculations were done using ArcGis 10.3 nearest neighbor analyses found in the spatial analyst package. The distance between each species was calculated for each transect using all data collected for the transects. Nearest neighbor analysis indicated if species were clumped into herds. Once herds were identified, results provided the mean distance between herds and the percent likelihood that the observed distribution patterns were random.

**River morphology.** Species richness and abundance were correlated with river morphology using the ArcGis 10.3 Spatial Analyst program based on the body size for each species. Satellite imagery was used to identify deep, moderately deep and shallow channels, wet and marshy floodplains and the slope of channel banks. Channel heterogeneity was
measured based on the number of meandering channels, braided channels, the depth of channels, presence or absence of alluvial fans, amount of marshy (fen, bog, or deep mud) area around channels, and amount of dry land adjacent to channels. We then divided each section of the research site according to how many channel and substrate types were present. We graded these on a scale of 1-5, low heterogeneity, low-moderate, moderate, high and very high heterogeneity. We then used the ArcGis 10.3 spatial analyst tool to correlate species geographic locations, richness, and abundance with river morphology using a correlation model found in the ArcGis 10.3 spatial analysis package. We subsequently ran chi-squared test to determine if our results were random.

**Temporal patterns.** Temporal patterns were also mapped using ArcGis 10.3 allowing for more detailed examination of species assemblage patterns over time. We searched for patterns of richness and abundance within landscapes along with habitat associations, assemblages (nearest neighbor), and abiotic drivers, such as time of day, water, temperature, and season.

**Species richness and abundance.** We used a Chi-square test to determine if species richness and abundance were greater within specific research areas. As the enclave was inaccessible for several months due to flooding, fewer transects were driven in this portion of the research site than others. We therefore reduced the number of datasets analyzed for the other three portions of the research site to match the number driven in the enclave. Similarly, some parallel transects were longer than others so we reduced the length of parallel transects to equal 8 kilometers each.

Dataset analysis was complicated by the frequency of zeroes recorded when no species were present. A large number of zeros has the possibility of confounding the chi-
square test. We therefore ran the test in three different ways: first with the zeros included, which yielded a p value of $<2.2\times10^{-16} = 0.00001$. In subsequent tests we called our zeros 1 or 2 and increased our counts by one or two respectively. The p values for both of these tests were the same as the first test where zeros were included. We therefore concluded that the zeros were not confounding the dataset and that the p value could be trusted. Each dataset had over 64 degrees of freedom, which may indicate that our sample size was large enough to be unaffected by the presence of zeros. Regardless, it is obvious from the data that richness and abundance levels in some of the study areas are significantly higher than in others.

**Results**

**Nearest neighbor.** Herbivore herds with an average of 26 individuals spaced themselves at a mean distance of 23.5 meters apart. In Linyanti, herds were clustered 21 meters apart, in Rann 26 meters apart, in the Enclave 24 meters apart, and 23 meters apart along the Chobe Riverfront. All nearest neighbor analyses indicated a less that 1% chance that the spatial patterns were random and standard deviations were less than 30 percent for all sites. In addition, our spatial analyses indicated that the majority of herds are clumped within 2 km of the river during the dry season.

**River morphology correlations.** We found that in sites with low channel heterogeneity (consisting of only a grassy floodplain) that species richness and abundance were both the lowest, with an average of four species using the floodplain on any given day during the dry season (Figure 1.2). Only Zebra (Avg. 30/day), baboons (Avg. 14/day), waterbuck (Avg. 12/day), and Sable (Avg. 1/day) seemed to frequent these short grass floodplains, with waterbuck occurring where tall reeds dominated. Richness was similar in
the floodplain with low-moderate heterogeneity, which included floodplains inundated with shallow water. Here an average of five species could be seen in the floodplain on any given day during the dry season (Figure 1.2), but the composition of species was quite different. Zebra were no longer present and impala (62), followed by baboons (45), then lechwe (28) dominated the landscape, other species included the occasional kudu (6) and giraffe (5).

Species richness more than doubled (5-13 sp) in moderately heterogeneous landscapes where one deep channel and floodplains with bogs, fens or just deep mud began to dominate. Here an average of 13 species could be seen on any given day in the dry season (Figure 1.2). Baboons (233) dominated the landscape followed by elephants (73), then giraffe (26) and buffalo (22), followed closely by waterbuck (21) and kudu (20). However, abundance levels remained low compared to sites with high and very high heterogeneity.

In floodplain landscapes with high heterogeneity (consisting of one deep channel and a few shallow braided channels, including some grassy floodplains) abundances for many species doubled or tripled again and richness increased by two species (Figure 1.2). Here, baboons (371) continue to dominate based on number, but are followed closely by elephants (309) who dominate based on biomass. Zebra (109) show up again in areas with short grass floodplains close to shallow water, followed by Kudu (75), lechwe (68), buffalo (62) and giraffe (53). In addition, hippopotamus only begin to show up where channel heterogeneity is high. In the portion of the floodplain with the highest heterogeneity (very high), species richness increased insignificantly by one species (Figure 1.2), however species abundance more than doubled, from 1131 individuals to 2660. Here, elephants (669) and buffalo (658) dominate the landscape, followed by lechwe (320), baboons (271), impala (238), and kudu.
Sable that had been relatively rare previously, could be seen in abundance. Species not seen in this landscape were zebra, roan and steenbuck.

Body size seemed to have the greatest impact on the degree of floodplain heterogeneity preferred by species (Figure 1.1 & Table 1). We ran a chi-squared test to determine if our body size correlations occurred because of chance or if there was a pattern. Chi-squared tests produced an X-squared = 2021.754, 64 degrees of freedom, and a p-value < 2.2e-16, indicating that our results were not a result of chance. Species with the largest body mass (750-4000kg; Log 3-3.7) only used portions of the floodplain with deep channels (figure 1.1). Zebra, horse antelopes (*Hippotragus*), such as sable and roan, and wildebeest were found mainly where short grass floodplains existed and avoided floodplains with shallow water and moderate heterogeneity consisting of bogs and deep mud (Figure 1.1). Smaller bodied species used all areas, but preferred channels with the most heterogeneity (Figure 1.1). This pattern was similar for most species, all of which tended to congregate in areas where channel heterogeneity was very high. While used by most species, areas with bogs, fens and deep mud had low abundance levels (Figure 1.1).

**Temporal movement patterns.** We estimated that if species were within 500 meters of the river that they were either moving toward the wetland to drink, away from the wetland after drinking, or they were at the water (Figure 2).

Time of day seemed to have the greatest overall effect on when species access water. Two peaks in species abundance during the hot-dry season occurred: one between 9:00 and 10:00 am and another between 4:00 and 6:00 pm (Figure 2). These time intervals represent the period just before peak daytime temperatures, which occur between 12:00 and 3:00 pm and the period immediately preceding daily temperature highs. The pattern is similar during
the cold-dry and cold-rainy seasons with varied smaller peaks throughout the day and a large peak between 3:00 and 5:00 pm in the cold-rainy season (Figure 2) –roughly the hottest time of day –and a peak between 2:00 and 4:00 pm in the cold-dry season (Figure 2). This pattern is reversed during the hot-rainy season with peaks occurring during the hottest time of day and abundance levels being significantly lower throughout the season except in the evening. During the hot-rainy season the sharp increase in species movement to and from the wetland occurs at 5:00 pm (Figure 2) instead of at 3:00 pm.

Species assemblage patterns in space and time become more defined as the scale of observation is narrowed. While large species access water points independently (elephants and buffalo), smaller species access water collectively. An even greater distinction in temporal patterns of use can be seen when species movements are plotted at 15-minute intervals (Figure 3 and 4). Buffalo, elephants, and giraffe rarely access water together (Figure 3). Conversely, the data suggest that smaller species (300-50kg) access water collectively (Figure 4). The tight temporal variations in species use of the wetland combined with the intricate spatial variation suggests grouping of species, and in some cases guilds, with specific behaviors for accessing the water resource in both space and time.

**Species richness and abundance.** The Chobe Riverfront is adjacent to the village of Kasane on its east side and the Chobe Enclave on its west side. Species abundance is highest in this portion of the study area (Figure 5). Linyanti was adjacent to the hunting area, RANN, on its east side, during the time of the study. Species richness was highest in this portion of the study area and abundances were moderate when compared to other sites. We predicted that species richness in the Enclave would be lower than in other areas due to competition for resources with humans and livestock, and predation by humans. This appears to be the case
Likewise, species abundance is low as well (Figure 5). Within the community hunting concession, Rann, species abundance was the lowest, but richness was equivalent to the Riverfront (Figure 5).

**Discussion**

The fact that species access water at specific intervals during each season provides an opportunity for humans accessing the same water points to avoid the riverfront and wetland during certain times of day. It also informs farmers of the best times to monitor fields that are close to water (malapa farms) and may be opportunistically raided by herbivores. It does not account for night raids by elephants. Providing planned access points for the largest and most problematic species, such as elephants and buffalo, may further diminish human-wildlife conflict. By maintaining open access to areas with flat to moderate bank slopes and areas with a high degree of channel heterogeneity that include deep channels, humans may be able to steer large herbivores to predictable water points along the floodplain.

Species abundance and richness were highest in areas where channel heterogeneity was high. The largest and most problematic of species (elephants and buffalo) prefer channel morphology that includes deep channels and bank slopes of 0-20 percent, suggesting that it may be possible to provide specific water access points that large herbivores will prefer. Other studies have indicated that elephants prefer slopes of 0-9 percent (Matawa et al. 2012). The difference in our results may occur because other species were included in our analysis or because data from others studies likely differed in spatial or temporal resolution. It was also shown that all species are clumped within two to four kilometers of the river and floodplain during the dry season, indicating that, while some species may be less water dependent than others, most maintain a close proximity to water, which suggests that
movement corridors placed outside of this range may not be optimally utilized by large herbivores.

As was shown by Selier et al. (2015), habitat modeling for large herbivores is often not done at a scale relevant to conservation decision-making. Wildlife corridor planners should use these types of daily movement data when delineating optimal movement paths for wildlife. Similarly, foraging areas can be mapped in order to delineate optimal paths that species will take when moving between foraging sites and watering points.

Selier et al. (2015) and Hopcraft et al. (2014) have shown that large herbivores, and in particular elephants, trade-off between predation risk and access to high quality food. In both studies, herbivores avoid humans, and to a larger degree hunting areas, even when high quality resources are available. On the other hand, they sometimes ignore other predators, such as lions to access high quality forage (Hopcraft et al. 2014; Selier 2015). Similarly, our data has shown that human presence, their livestock, and hunting appear to be acting as deterrents to wildlife based on the low levels of both species richness and abundance in hunting and human use areas. We suggest that this phenomenon can be used to control HWC. If optimal movement corridors are created for large herbivores, hunting in well-planned buffer zones around farming areas and villages could decrease HWC.

However, our analysis on species richness and abundance in the various research sites may have been confounded by the fact that floodplain heterogeneity was very different among the different sites. Some sites (Enclave and Rann) consisted of multiple degrees of heterogeneity and were split up into smaller categories when we analyzed the data based on floodplain heterogeneity and the body size of animals. It may therefore be important to conduct new studies that include areas with similar degrees of floodplain heterogeneity.
consisting of different management regimes, such as: hunting, protected area, farming and ranching.

Human-wildlife conflict is likely to persist if humans continue to build within the critical habitat of large herbivores. A means for mitigating conflict incidents must be identified that does not include restricting the movements of large herbivores, but rather modifies the movements of humans. In some cases appropriately located underpasses and over passes may facilitate wildlife migrations through urban areas. We can reduce our impact on wildlife due to land-use change by having planners consult with the relevant wildlife biologist when developing linear infrastructure projects, such as roads and railways, and when developing plans for new cities and villages. We also suggest that humans worldwide can modify their behavior to mitigate conflict by accounting for the movement and assemblage needs of large herbivores and other migratory species.

We recommend that additional data be collected on an annual basis in order to provide more certainty concerning these daily movement patterns. With subsequent studies it is expected that more detailed patterns will emerge to allow humans to further predict the needs and movement requirements of large herbivores. For instance, it is also important for humans to predict in which inland habitats dangerous species, such as buffalo, will be congregating when not accessing water. For buffalo it is likely grasslands and for elephants it is likely mopane stands in the dry season and *Acacia*, *Combretaceae* and *Terminalia* species in the wet seasons. It is also probable that species movements to water are more correlated with temperature than time of day, but more data are required to verify this assumption. With global warming these time intervals may vary and species may become more restricted by the intervals available to them for watering.
Our results have broad implications for wildlife conservation. They provide an important view of the daily assemblage patterns, habitat needs, species interactions, and movement requirements of the region’s mega-herbivore super-guild when accessing water. If species movements are somewhat predictable and strongly correlated with water access types, this may allow humans to respond in ways that can begin to reduce conflicts. It makes available the data needed for habitat modeling to create movement corridors for large herbivores at a scale that allows species to move between various resources within hourly and daily time intervals in the dry season. It will also assist village planners in restricting development to areas that will not impact the movements of large herbivores.

Acknowledgments

Grateful thanks are expressed to the Botswana Department of Wildlife and National Parks for providing permission to carry out this study. We would also like to thank Kathy Alexander and the Center for African Agriculture, Conservation and Land-Use for their support in helping us to carry out this study and the Fulbright Program for providing funding. We also thank Golden Kamedi for his help and support.
References


Table 1: Average body size of large herbivores and 3 other species. For sexually dimorphic species, the average between male and female body size is given. The above is a list of all the species monitored throughout the study. Baboons, vervet monkey and lion are only included in diversity and abundance counts not associated with movement patterns.

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<th>Scientific Name</th>
<th>Common Name</th>
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<th>Avg. Male KG</th>
<th>Average/KG</th>
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Figures

**Figure 1:** Above: shows average number of individuals seen adjacent to or within floodplains with varying degrees of heterogeneity. Data from the dry season prior to complete inundation of the floodplain was used for all sites. Low heterogeneity (floodplain consisting of short grass and an occasional waterhole) = light green. Low–moderate heterogeneity (Floodplains with shallow water) = light blue. Moderate heterogeneity (one

**Figure 2:** Species Abundance by Floodplain Heterogeneity

Species Use of Wetland Types Based on Body Size

Species Richness & Abundance by Degree of Floodplain Heterogeneity

Figure 1: Above: shows average number of individuals seen adjacent to or within floodplains with varying degrees of heterogeneity. Data from the dry season prior to complete inundation of the floodplain was used for all sites. Low heterogeneity (floodplain consisting of short grass and an occasional waterhole) = light green. Low–moderate heterogeneity (Floodplains with shallow water) = light blue. Moderate heterogeneity (one
large channel with fen, bog or extensive and deep muddy floodplain) = mossy brown. High heterogeneity (Deep channel adjacent to a few braided shallower channels, grassy or inundated shallow floodplain) = dark green. Very high heterogeneity (Deep channel adjacent to many braided shallower channels, grassy, and inundated shallow floodplain) = dark blue. **Below: bar graph shows average number of species seen adjacent to floodplains with different degrees of heterogeneity during the dry season.** Pie chart: shows abundance of each species within floodplain types during the dry season. Results of our analysis yielded X-squared = 2021.754, df = 64, p-value < 2.2e-16 =< .00001.

**Figure 2: Abundance of all species accessing water according to time of day.** The above graphs show the abundance of all species combined accessing the water throughout the day during a given season. Time intervals and abundances are aggregated for each season to show patterns.
Figure 3: River access times for herbivores over 80kg. This figure shows the number of individuals accessing the river by time of day for three of the largest herbivores, buffalo, elephants and giraffes at the same location. Data are shown at 15-minute intervals illustrating that these three species rarely access water points at the same time of day.
Chapter 3: Hunting Bans Can Precipitate Illegal Hunting and Increase Land-use Change, Which May Be Exacerbating Wildlife Declines in Botswana

Abstract

Hunting bans can have the unintended consequence of exacerbating biodiversity loss. Bans increase illegal hunting and land-use change. Using Botswana as a case study, we provide an example of how lack of local ownership in safari and trophy hunting industries has led to the establishment of secretive bushmeat markets. We explore the potential drivers of species loss and illustrate how the hunting ban has: led to loss of local livelihoods magnifying the need for illegal hunting; compelled people to obtain more livestock to increase their incomes, and displaced rural people leading to land-use change. We show how land-use change increases illegal hunting and human-wildlife conflict, fragments habitat, and blocks migratory routes, causing additional wildlife declines. We suggest that mitigating other drivers of species loss will have a more positive impact on biodiversity than prohibition of hunting. We provide an example of a method for valuing wildlife resources that supports rural communities and increases biodiversity.

Introduction

The illegal hunting of wildlife is one of the biggest threats to biodiversity worldwide (Schipper et al. 2008; Roe 2015; TRAFFIC 2016) because the illegal sale of wildlife and wildlife parts is the third most lucrative illegal market in the world (Ayling 2013). The poaching of elephants (Loxodonta africana) has tripled in Africa in the past five years and approximately 30,000 elephants are killed each year for their ivory (TRAFFIC 2016; UNEP et al. 2013). This increase in illegal hunting has led to aggressive conservation programs
aimed at controlling poaching that often include hunting prohibitions (Namtgail et al. 2009; MEWT 2013; Duffy 2016).

Hunting bans have been shown to have positive impacts when specific threatened and endangered species are targeted; as was the case with the ocelot (*Leopardus pardalis*) and jaguar (*Panthera onca*) (Caso et al. 2008; Paviolo 2015). However, research conducted over the past decade on the effectiveness of hunting prohibitions and their impacts on rural communities has shown that, on every continent and in all cultures, complete hunting bans can lead to local rebellions and increased poaching, whereas highly regulated hunting is usually supported and even enforced by communities that live closest to wildlife (Ayling 2013; Von Essen et al. 2014). Similarly, it is important to ensure that conservation strategies designed to reduce poaching do not effect additional species loss through unintended impacts. Hunting bans can destroy the livelihoods of rural people. As a solution, conservation programs often develop alternative income generating mechanisms that can unintentionally lead to land-use change and increased habitat fragmentation.

In 2010, results from an extensive aerial survey indicated a population decline of more than 60 percent for many species (S2 Table 1) in Northern Botswana (Chase 2011). These data compelled the government of Botswana to issue a complete ban on hunting in an attempt to prevent further loss of wildlife (MWET 2013). However, a recent report by Rogan et al. (2015) has shown that legal hunting off-take in Botswana remained far below the intrinsic growth rate of most species (Figure 1). This suggests that factors other than legal hunting may be the primary drivers of species declines in Botswana.

In 2012 and 2014 Botswana held wildlife workshops and assembled working groups to address that very notion. Participants (which included: Okavango Research Institute (ORI)
Scientists, The Department of Wildlife and National Parks (DWNP) biologists and researchers from around the world) agreed that additional research was needed to identify the drivers of wildlife declines. Veterinary fences (used to separate livestock from wildlife), changes in hydrology and land-use change emerged as the impacts most likely to have caused species loss (DWNP 2012a).

In this paper we explore how the hunting ban may be having unintended impacts on wildlife and people through loss of local livelihoods which is not only creating negative sentiments toward wildlife but pushing local people toward income diversification that includes illegal hunting and increased land-use change. We suggest a system of valuing Botswana’s wildlife resources as a way to gain investments to fund the infrastructure developments needed for rural communities to continue living sustainably. We evaluate the other potential drivers of species loss and suggest that measures to mitigate those impacts (land-use and climate change) will have a more positive impact on both people and wildlife than prohibition of hunting.

The Status of Wildlife in Northern Botswana

Botswana is home to one of the most abundant wildlife populations in Africa, due mainly to its progressive conservation strategies, and the fact that 40 percent of the country is devoted to conservation (Fynn & Bonyongo 2010; Chase 2011). Its Community Wildlife Management Area (WMA) system has provided a mechanism for rural communities to enter into the photo safari and trophy hunting industries, which is usually done through joint ventures with foreign safari companies that are required to employ a certain number of people from the community.
Botswana is 84 percent desert and its vast wildlife populations are mainly restricted to the Okavango Delta and Chobe National Park where water is abundant (Figure 5). For this reason, aerial wildlife surveys between 1993 and 2004 have mainly focused on these regions. Between 2010 and 2012 Botswana increased the accuracy and extent of its aerial surveys. The 2010 survey conducted by Elephants Without Borders (EWB) covered 73,478 square kilometers, focused on 21 mammals, as well as cattle, and included 10 percent more coverage than previous surveys (Chase 2011; DWNP 2012b). In 2012, DWNP conducted a subsequent survey that covered the entire country and focused on 29 species, including cattle, sheep, and goats (DWNP 2012b). Due to the discrepancy in survey size, population estimates are only given for species within the Okavango Delta, which includes the Moremi Game Reserve (MGR) and the surrounding Ngamiland District, and for the Chobe-Linyanti River and Wetland System, which includes Chobe National Park and the surrounding Chobe District.

**Population estimates.** The 2010 survey revealed population declines of 60 percent or more for 11 species in the Okavango Delta between 1993 and 2010 (Chase 2011). While some species’ population declines were statistically significant, others have fluctuated greatly and were not statistically significant. Buffalo (*Syncerus caffer*) exhibited a drastic decline in the MGR between 1999 and 2010 (Figure 2a), but showed an increase outside of MGR in the surrounding Ngamiland District between 2004 and 2010 (Figure 2b). Elephant populations inside the MGR and Ngamiland remained relatively constant (Figures 2a-b), but almost doubled in CNP and the Chobe District between 1996 and 2010 (figure 2c-d). Tsessebe (*Damaliscus lunatus*) populations declined by 87 percent in MGR (S1 Figure 4e) and wildebeest (*Connochaetes taurinus*) populations declined by 90 percent in MGR and
Ngamiland (S1 Figure 4b & 5b). Zebra (*Equus quagga burchellii*) populations have been consistently increasing in all regions except Ngamiland since 2004 (S1 Figures 2b, 3b, 4b & 5b) and eland have exhibited wide population fluctuations in all regions from year to year (S1 Figure 2c, 3c, & 5d). These heterogeneous population dynamics indicate that the causes of wildlife declines are both regional and diverse and compel wildlife to shift their resource use patterns according to regional drivers.

This is further illustrated by the fact that in 2012 changes in wildlife abundances were quite different. Most species in MGR and Ngamiland exhibited population increases, with the exception of lechwe (*Kobus leche*), sable (*Hippotragus niger*), and hippopotamus (*Hippopotamus amphibious*) (S1 Figures 4 & 5). In CNP, most species declined in 2012, with the exception of zebra, wildebeest, elephants, and roan (*Hippotragus equinus*) (S1 Figure 2). However, when populations are estimated for the entire country only hippopotamus, lechwe, sable and tsebebee showed declines in 2012 (DWNP 2012b; S2 Table 1). Population increases for most species were drastic (increase of: 79,000 elephants, 21,000 buffalo and 63,000 impala (S2 Table 1) indicating that: 1) the 2012 survey estimates were much higher due to the increased coverage of the aerial survey; 2) elevated precipitation caused drastic population increases; or 3) species migration to and from Botswana is important (although, it should be noted that some species exhibiting drastic population increases, such as impala (*Aepyceros melampus*), do not migrate).

**Potential drivers of species declines.** As Chase (2011) pointed out, wildlife declines in Northern Botswana coincided with a 20-year drought that began in 1981 and persisted until 2010. In Ngamiland, precipitation in 2010 and 2011 was the highest (900mm) recorded in 20 years (Botswana Dept. of Water Affairs 2011; Chase 2011) and may have facilitated
population increases in many species in 2012. Increased precipitation may have also
compelled many species to shift their resource use patterns, making population estimates
difficult. For instance, the sharp decline in hippopotamus populations in CNP in 2012 (S1
Figure 2f) could be a result of increased flooding. Hippopotamus may have been less visible
under deep water, and likewise, lechwe (*Kobus leche*) which are extremely water dependent
may have shifted their resource use patterns to avoid deep water (S1 Figure 2f).

Governed by precipitation in Angola and Zambia, the hydrology of the flows that
reach Botswana in the dry season has also changed (DWNP 2012a), with flows between
2009 and 2012 also being the highest recorded in the past 20 years (Chase 2011; DWNP
2012b; Botswana Dept. of Water affairs 2012). As O’Connor pointed out in the “Future of
the Okavango’s Wildlife” (FOW) Workshop (2012), sediment that is carried with
floodwaters is backfilling channels and shifting most channels in the Okavango Delta to the
east (DWNP 2012a). Such shifts change the structure of vegetation and the resource-use
patterns of wildlife that depend on it (DWNP 2012a). Meanwhile, the Savuti Channel, which
was dry for 25 years, is now running and provides water where high quality forage (*Acacia
definans*; *Acacia spp.*) is also available to elephants, causing them to be less abundant in the Okavango Delta
and Linyanti regions (Chase 2011; Teren & Owen-Smith 2010). This indicates that aerial
surveys conducted within the borders of Botswana may be overlooking populations that have
shifted their resource use patterns outside of the country.

Botswana’s elephant population continues to increase and some researchers have
asserted that competition for resources with elephants could be impacting other species. One
study of the Okavango Research Institute has shown that, where the Southern Buffalo Fence
was removed, elephants have simplified the landscape and caused a general decline in
biodiversity (Cassidy DWNP 2012a). There is, however, no conclusive evidence to support this and other studies have indicated the opposite (Arsenault & Owen-Smith 2002; Skarpe et al. 2004).

On the other hand, veterinary fences erected in the 1960s to keep cattle from contracting hoof and mouth disease (HMD) from wildlife have been strongly correlated with wildlife declines (Chase & Griffin 2009, 2011; Chase 2011). Fences blocked critical wildlife movement corridors between the Makadikadi Game Reserve (wet season habitat) and the Okavango Delta (dry season habitat) (Chase 2011; Cushman et al. 2005; Fynn et al. 2014) and between the Moremi Game Reserve and the Caprivi Strip (Chase 2011; DWNP 2012a). These barriers caused the die-off of many species that were cut off from their dry season habitats (Chase 2011; Fynn et al. 2014). The removal of the Southern Buffalo Fence (between the Makadikadi and the Moremi Game Reserves) has resulted in zebra population increases only 5 years after its removal (Chase 2011). Livestock also compete with wildlife for resources and have been shown to be one of the strongest deterrents to native African herbivores through resource competition, second only to hunting (du Toit & Cumming 1999; Selier et al. 2015).

Land conversion to agricultural fields and new settlements are also restricting wildlife movements and making key habitat for many species inaccessible (Chase 2011; Gureja et al. 2014). Selier et al. (2015) showed that large herbivores in Southern Africa trade off between accessing high quality forage and avoiding human disturbance. This type of trend has been shown in many studies across Africa (Verlinden 1997; Winterbach et al. 2014; Selier et al. 2015) and is the number one driver of biodiversity loss worldwide (WWF 2014). In addition, humans living in close proximity to wildlife almost always leads to human wildlife conflict,
which exacerbates wildlife declines and undermines human sensitivity toward wildlife (Elliot et al. 2008; WWF 2014). In Botswana, lack of appropriate collaboration between agriculture, housing, veterinary and environmental authorities leads to the placement of farms, veterinary fences, and other human developments in critical wildlife habitat and migratory paths (Gureja et al. 2014).

Selier et al. (2015) demonstrated that hunting had the biggest avoidance impact on large herbivores followed by livestock. Large herbivores know when and where hunting is occurring and access resource use areas when threats are low (Selier et al. 2015). Species displacement as a result of hunting is only temporary and does not contribute to the national decline of species. When the hunting season is completed animals return. On the other hand, livestock grazing is constant and displaces wildlife indefinitely.

Trophy hunting, if done according to regulations, should not have an impact on wildlife populations except through secondary impacts to population genetics. In Botswana, hunting quotas are set at a fraction of species’ growth rates and it is only legal to hunt males, which should result in no impacts to future growth rates, as males are able to mate with more than one female (DWNP 2012a). The intrinsic growth rate of species is calculated based on population size and the number of fecund females in a population (Table 1). Nevertheless, most species are exhibiting negative rather than normal growth rates (S1 Figure 1a-d) indicating that drivers other than legal hunting are causing wildlife declines.

Chase (2011) calculated the growth rates for the majority of species recorded during the 2010 aerial survey based on previous and current fluctuations in population size. We then calculated the expected increase or decrease in population size for 2011 and 2012 based on those numbers. We compared our calculations with actual population changes and
discovered that the predictions were not very accurate (S1 Figure 1a-d; S2 Tables 3-6). In Ngamiland, species declined less than expected between 2010 and 2012, with the exception of buffalo, which increased less than expected (S1 Figure 1b). In the Chobe District, species appeared to decline instead of increase as expected, with the exception of wildebeest and buffalo whose populations increased more than expected (S1 Figure 1d). Zebra population increases were as predicted by Chase (2011) in the Chobe District (S1 Figure 1d). In MGR most species increased more than expected with impala and zebra increasing well above the expected level and warthog, buffalo and hippopotamus declining below expected levels (S1 Figure 1a). If legal hunting were the driver of species declines we would see a clear decrease in species populations according to predicted growth rates. The fact that we do not indicates that fluctuations in species abundances are driven by other variables, including illegal hunting.

Researchers have pointed out that, in Botswana, legal hunting is not regulated well enough to ensure that quotas are adhered to, and illegal hunting is often concealed in legal hunts (DWNP 2012a). Similarly, poachers use legal hunts as a way to identify areas where poaching will not be detected (DWNP2012a; Gureja et al. 2014). On the other hand, studies from all parts of the world have shown that hunting prohibitions usually lead to increases in poaching that exceed legal hunting quotas and target females as well as males, which leads to population declines (Ayling 2013; Von Essen et al. 2014). When reasonable hunting restrictions are put in place and the benefits of hunting shared with communities, former poachers often become the most effective anti-poaching guards. This usually leads to drastic reductions in poaching and increased wildlife populations, which is the case next door to Botswana in Namibia (Kahler & Gore 2015; Clark 2015). Currently, take from illegal
bushmeat consumption in Botswana accounts for as much as 4 percent of some species’ populations (Rogan et al. 2015).

Data gathered from anti-poaching patrols for 2008-2014 recorded approximately 500 incidents, and suggest that the majority of species killed by poachers in Botswana are elephants and Kudu (*Tragelaphus strepsiceros*), followed by impala and rhino (DWNP 2012; S2 Table 2). A poaching study presented by Kai Collins at the 2012 FOW workshop, revealed 204 poaching incidents in one wildlife management area, none of which were recorded by anti-poaching units. This study indicated a very different demographic of species being poached, the majority being buffalo, followed by lechwe, then impala (DWNP 2012a). Similarly, a recent study conducted by the UNDP on illegal bushmeat hunting in the Okavango Delta, that relied on community surveys and undercover interviewers, has revealed that as many as 2000 illegal bushmeat hunters are active in the Delta harvesting an average of 357,250 kg of meat annually (Rogan et al. 2015).

Nevertheless, it is clear that other factors are causing more significant wildlife declines. Changes in rainfall and hydrology likely lead to population fluctuations that are a result of die-offs and wildlife shifting their use patterns. These shifts require animals to move between important resource use areas that, in some cases, have been blocked by veterinary fences, farms and new developments. Wildlife is also significantly displaced by livestock and temporarily displaced by hunting. In many parts of the world, including Africa, land-use change has been shown to be a driver of illegal hunting and HWC (Haines et al. 2012; Radovani et al. 2014; Selier et al. 2015). Illegal hunting, legal hunting and HWC are likely exacerbating wildlife declines resulting from other drivers.
Thus, without conclusive evidence indicating the main driver of species declines, three clear mechanisms for mitigating wildlife loss emerge: 1) Reduce land conversion and habitat fragmentation in critical habitat areas and migratory corridors; 2) remove veterinary fences in critical migratory corridors; and 3) identify and mitigate the root cause of illegal hunting in Botswana.

**What causes people to hunt illegally?**

Despite the limited evidence that legal hunting is the driver of species declines in Botswana, a hunting ban has been implemented and its potential impacts need to be addressed. Illegal hunting has been identified as ‘the most systematic and formalized explanation of defiant behavior in the literature’ (Curcione 1992). Documentation of community reactions to hunting bans dates as far back as eighteenth-century England, where poaching outlaws formed gangs in protest of hunting prohibitions and were called “the blacks” by wealthy landlords (Thompson 1975). Since then, many studies have documented social protests that arise when hunting bans are imposed on a population whose subsistence and or livelihood depends on it (Carruthers 1995; Mackenzie 1988; Garland 2008; Roe 2008; Robbins et al. 2009; Groff & Axelrod 2013; Fischer et al. 2013).

Duffy et al. (2015) clearly illustrated how the origins of hunting bans in Africa can explain why communities are still resistant to hunting regulations. Colonial governments in most of Africa culled huge numbers of large herbivores to make space for farming and to reduce human-wildlife conflict (Kangwana 1995; Hoare 1999). These same governments then restricted the rights of Africans to hunt in order to protect the trophy hunting and safari industries (Carruthers 1995; Mackenzie 1988; Garland 2008; Roe 2008b; Robbins et al. 2009; Fischer et al. 2013; Duffy et al. 2015). Naturally, communities rebelled against these
unjust hunting bans, which were implemented to assist colonists in amassing wealth and security.

It has been shown that people who engage in illegal hunting do so when economically and socially oppressive laws enable them to justify their actions to such a degree that the guilt associated with the crime is eliminated (Curcione 1992; Eliason & Dodder 1999; Jones et al. 2008). Communities have also been shown to increasingly protect illegal hunters from authorities (Von Essen et al. 2014). When outsiders begin poaching the sentiment of communities often changes and outsiders are viewed as poachers and thieves. However, this only persists as long as the livelihoods of the affected community members do not suffer as a result of hunting prohibitions (Von Essen et al. 2014; Duffy et al. 2015). When incomes are lost, those same community members often feel justified in assisting foreigners in illegal hunting.

Secretive illegal bushmeat markets currently exist in most villages in the Okavango Delta where meat is sold and traded among villagers (Rogan et al. 2015). An illegal hunter in this region typically harvests an average of 399 USD worth of meat a year (Rogan et al. 2015). These same hunters commonly have more livestock than the average community members and are usually employed (Rogan et al. 2015). As Rogan et al. (2015) point out, this indicates that illegal hunting in the delta is a livelihood mechanism and not a subsistence mechanism. While community members have enough livestock to sustain their families, they prefer to eat bushmeat than to slaughter their own animals. Hunting allows families to raise more cows, which increases their status in the community, but it also leads to land-use change as livestock numbers increase.
The income families gain from the bushmeat trade is far less than the income one family member would receive when working for the safari industry or even from a minimum wage job in the agricultural industry (Rogan et al. 2015). To understand why community members prefer to hunt and supplement their incomes with bushmeat rather than obtaining a job, one must understand what poverty actually is (Duffy et al. 2015). An analysis of what constitutes poverty was done by Sen (1999) and includes an inability to define one’s future, lack of power, lack of prestige, not being heard and not being able to control one’s day to day activities.

Wilkinson and Pickett (2009) argued that once abject poverty is alleviated, the most socially and economically equal communities are the happiest, healthiest, and experience the least amount of crime and stress. Botswana is noted for being one of the most socially and economically equal societies with a minimal amount of poverty (Wilkinson & Pickett 2009).

This indicates that Batswana will be less likely to place themselves in a situation where they will feel unequal to the people around them (Wilkinson & Pickett 2009) or where they cannot define their own futures.

When an individual takes a job, especially an unskilled job, they are subordinate to the managers and owners of that business and obligated to be at one place every day. Thus, making less money and maintaining one's independence may be more appealing than obtaining a job. The photo safari and trophy hunting industries create the same class disparity that colonialism did; wherein the only perceived avenue out of poverty is to work as a subordinate for wealthy foreigners. The hunting ban reinforces this sentiment by allowing wealthy landowners with fenced game ranches to continue trophy hunting (MEWT 2013) while taking those rights away from local people.
Secretive bushmeat markets are an indication of social discontent that is exacerbated by the hunting ban. Recent articles published where community members are speaking out against the hunting ban and even admitting to killing lions and other animals are indications that whole communities feel justified in breaking the law (Onishi 2015; Kgamanyane 2015). Anti-poaching data indicate that a large number of apprehended poachers are Botswana working in concert with foreigners (DWNP 2014; Supplemental data Table 4) suggesting that money is a motivator.

Thus, the trophy hunting ban is magnifying the class disparity between safari operators and local people. History has shown that this will only lead to increased social unrest (Wilkinson & Pickett 2009) and increased land-use change having a double negative impact on wildlife. Rather, communities need to be given greater ownership of the natural resources that surround them and regulations should be put in place that promote sustainability. In Namibia, where hunting is legal and regulated, and communities have been given control of their wildlife resources, species’ populations are consistently increasing (Kahler & Gore 2015; Clark 2015).

**Indigenous people and biodiversity.** Indigenous communities the world over have both safe-guarded and co-evolved with our planet’s biodiversity for millennia and currently inhabit 80 percent of the world’s most biologically diverse regions (Levin et al. 2001; Dowie 2009; GEF 2015). In many cases they have unintentionally protected or enhanced biodiversity through their sustainable use of resources (Levin et al. 2001; Dowie 2009). This is the case in Southeast Asia where fruit gardens managed by indigenous communities, who have been cultivating forests for 11,000 years (Moore et al. 2016), were positively correlated with higher biodiversity than the surrounding forests (Moore et al. 2016). On the other hand,
some communities have developed very intentional methods of protecting the biodiversity they depend on through taboos and other cultural customs (Levin et al. 2001).

It is also important that conservation professionals take the time to understand the social implications of their actions before intervening in the lives of rural indigenous people. Insisting that these communities engage in conservation projects and alternative livelihood practices can, in some cases, have an unintended effect. A study by Bare et al. (2015), that assessed the impacts of international conservation aid on deforestation, found that conservation projects were associated with higher rates of deforestation, most likely due to the displacement of community members, the introduction of power tools and guns, and loss of sustainable livelihoods. Good governance that included local people in sustainable livelihood development and provided land tenure to indigenous people was the only factor that moderated this impact (Bare et al. 2015).

This phenomenon can be observed clearly in Botswana where the resources rural people rely on have been impacted by modern interventions, such as veterinary fences and land-use change. These interventions added to the decline of wildlife populations, which led to a hunting ban that has had its most negative impact on the people who have neither caused the decline nor benefited from its drivers. These same people are now viewed as the rural poor, in need of handouts from the foreign-dominated safari industry. Lack of opportunities for these people to increase household incomes have led to the creation of secretive bushmeat markets that allow families to increase their livestock herds as a means of amassing wealth, which further displaces wildlife through competition for resources. In addition, the hunting ban has forced Bushmen to move away from rural villages where they relied on hunting and gathering or to obtain cows to feed their families.
As we push indigenous communities out of their traditional lands and away from their subsistence livelihoods, we are at the same time promoting sustainable development and reduced land conversion as the number one way to slow climate change and halt the ever-increasing mass extinctions occurring across the globe (UNEP 2010). The Rio Conference on Sustainable Development highlighted the need to “achieve sustainable development by promoting sustained, inclusive, and equitable economic growth… while facilitating ecosystem conservation” (UNEP 2010). However, no clear guidelines have been developed for how sustainable development can support indigenous people.

**Future Directions**

We suggest that, together with strong regulator policies, systems for valuing wildlife resources can be put in place that support rural communities and compel them to protect and increase their wildlife resources. This type of model gives rural communities the opportunity to define their own futures and rewards them for the ecological service they have provided to us all by protecting our wildlife for thousands of years.

Valuing wildlife resources also provides communities with an incentive to continue living sustainably. It is clear that modern services are needed in rural villages and wildlife resources can be used to leverage outside funding to create sovereign wealth funds, similar to Norway’s, that can pay for the infrastructure needed to provide services such as: solar power cooperatives, grey and sewer water filtration systems and recycling, as well as livestock fencing and HWC mitigation.

To estimate the value of wildlife in just the Ngamiland District of Botswana, we obtained the average price of game meat sold in South Africa and estimated the amount of meat one could gain from one individual within a species (elephant, kudu etc). Our estimates
were based on the amount of meat gained from goats, cattle and sheep after slaughter. This equaled approximately 50 percent of the animals' weight in all cases (SADAFF 2010; Bahta et al. 2013). We then calculated the value of one individual from each species based on its weight and game meat value (51 BWP/lb=4.5USD/lb; SADAFF 2010). We suggest that these numbers could realistically be doubled, or even tripled, given the added ecological, aesthetic, and cultural value these species hold.

However, the conservative estimates we provide illustrate our point clearly. The wildlife resource in the Ngamiland District of Botswana alone is worth over 1 billion USD (1,031,085,000 USD). The current livestock resource is worth 355,264,000 million USD (Table 2). If investments were made against wildlife services they would represent 8,451 dollars per person in Ngamiland, if there are approximately 122,000 people in the region (Gureja et al. 2014). Such numbers could compel community members to employ tactics that increase wildlife populations the same as they attempt to increase their livestock herds.

The valuing of wildlife can begin with nontangible cultural and ecological services that must be maintained, such as: wildlife’s contribution to the ecological health and resilience of the Delta; and its aesthetic and cultural value to local people and the safari industry. After a certain level of population increase has been achieved, profits can be measured based on the percentage of wildlife that can be harvested without decreasing populations or impacting ecological resilience. Investors can essentially buy that rate of increase and expect returns based on the profits made in the “legal” bushmeat trade, from sustainable trophy hunting, the safari industry and services (water, sewer, solar power and recycling) provided to local businesses from infrastructure investment. This will also
incentivize livestock owners to reduce the number of cows they own and slow land-use change instead of facilitating it.

Game meat prices in South Africa and other parts of Africa (average 50 BWP vs the current 20 BWP) should be used to set baseline sale prices for legal bushmeat markets with subsidies given to native residents based on their economic status and number of cows owned. People with more cattle should pay more for bushmeat. Similarly, higher prices and assistance with distribution chains to lodges and restaurants can incentivize the legal game meat and fisheries trade for current illegal hunters. To ensure that the proper take limits are being adhered to, DWNP and trained community participants can monitor these markets.

Conclusion

It is clear that legal hunting is not the main driver of wildlife population declines in Botswana, but that land-use change, climate change and illegal hunting may instead be main drivers. The hunting ban only magnifies the need and incentive for communities to hunt illegally and increase their livestock herds, which leads to land-use change. Globally, wildlife represents a source of wealth and security for rural and indigenous people and methods for valuing and distributing that wealth to communities need to be developed. These funds should then be invested in sustainable infrastructure that helps to protect wildlife and increases the quality of life for rural people.
References


TRAFFIC. (2016). Elephants and Ivory. www.traffic.org/trade


Tables

Table 1: Hunting off-take represents the proportion of the population that was legally hunted. The intrinsic growth rate represents the proportion of population increase or decrease. The percentage of growth rate represents the percentage of the intrinsic growth rate that was legally hunted.

<table>
<thead>
<tr>
<th>Species</th>
<th>Hunting off-take</th>
<th>Intrinsic growth rate</th>
<th>Percentage of growth hunted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Buffalo</td>
<td>0.025</td>
<td>0.166</td>
<td>15.1</td>
</tr>
<tr>
<td>Giraffe</td>
<td>0.042</td>
<td>0.138</td>
<td>30.4</td>
</tr>
<tr>
<td>Impala</td>
<td>0.048</td>
<td>0.398</td>
<td>12.1</td>
</tr>
<tr>
<td>Kudu</td>
<td>0.272</td>
<td>0.256</td>
<td>106.2</td>
</tr>
<tr>
<td>Warthog</td>
<td>0.083</td>
<td>0.381</td>
<td>21.8</td>
</tr>
<tr>
<td>Wildebeest</td>
<td>0.108</td>
<td>0.265</td>
<td>40.8</td>
</tr>
</tbody>
</table>
Table 2: Population estimates for individual species in Ngamiland. Those estimates are used to estimate the worth of each population of species based on how much meat could be harvested if all the wildlife were sold on the game meat market. Weight in Kg: provides the average weight of one individual within the population. Kg of meat: indicates how much meat can be harvested from one individual of that weight. Price per animal: indicates how much an animal of that weight can be sold for based on the average market value (4.5 USD per Kg) of game meat in 2016. The population estimate for that species is then given and the total worth of the population is then estimated based on price per animal x the total population in Ngamiland.

<table>
<thead>
<tr>
<th>Species</th>
<th>Weight KG</th>
<th>KG of meat</th>
<th>Price per animal</th>
<th>Nagmiland Population</th>
<th>Population worth/USD</th>
</tr>
</thead>
<tbody>
<tr>
<td>Elephants</td>
<td>3000</td>
<td>1500</td>
<td>$6,750.00</td>
<td>126000</td>
<td>$850,500,000.00</td>
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<tr>
<td>Buffalo</td>
<td>1000</td>
<td>500</td>
<td>$2,250.00</td>
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<td>62000</td>
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<td>Wildebeest</td>
<td>190</td>
<td>95</td>
<td>$427.50</td>
<td>13000</td>
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<td>Kudu</td>
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<td>85</td>
<td>$382.50</td>
<td>5600</td>
<td>$2,142,000.00</td>
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<td>30</td>
<td>$135.00</td>
<td>69000</td>
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<td>350</td>
<td>$1,575.00</td>
<td>5000</td>
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<tr>
<td>Eland</td>
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<td>280</td>
<td>$1,260.00</td>
<td>900</td>
<td>$1,134,000.00</td>
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<td>Lechwe</td>
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<td>39.5</td>
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<td>Sheep/goats</td>
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<td>50</td>
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<td><strong>Total wildlife value</strong></td>
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<td></td>
<td></td>
<td></td>
<td><strong>$1,031,085,000.00</strong></td>
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<tr>
<td><strong>Total livestock value</strong></td>
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<td></td>
<td></td>
<td></td>
<td><strong>$355,264,000.00</strong></td>
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<tr>
<td><strong>Value of 1 wildlife share</strong></td>
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<td></td>
<td></td>
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<tr>
<td><strong>Value of 1 livestock share</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td><strong>$2,912.00</strong></td>
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Figure 1: Intrinsic growth rates compared with actual hunting off-take in Ngamiland. (Data from Rogan et al. 2014). The red bars represent the percentage of the population that was hunted legally in Botswana. Blue bars represent by what percentage the population is growing. When red bars are shorter than blue bars that indicates that hunting rates are below growth rates.
Figure 2: Population estimates for elephants & buffalo in all regions. (Data from DWNP 2012b & Chase 2011). Elephants are represented by circles with blue lines and buffalos by triangles with red lines.
Figure 3: Population estimates for Zebra & Wildebeest in all regions. (Data from DWNP 2012b & Chase 2011). Zebra are represented by dark purple lines and multicolored boxes and wildebeest by lavender lines and empty boxes.
Figure 4: Population estimates for giraffe & eland. (Data from DWNP 2012b & Chase 2011). Eland are represented by upside down by brown lines with triangles and diamonds with orange lines represents giraffe. No data is given for MGR because eland do not occur in the Moremi Game Reserve.
Botswana districts and park boundaries

Figure 5: Map of Botswana.
Supplemental Data
Figure S1: Expected Growth Rates for 2011 & 2012 based on growth rate calculations done by Chase 2011 compared with actual growth rates derived from population changes between 2010 and 2012. Growth rates were calculated using the logarithms of population estimates over time to create a linear regression model, which yielded slopes representing growth over time (Chase 2011).
Figure S2: Population estimates for Chobe National Park 1993-2012, data obtained from DWNP 2012b and Chase 2011. Species are coupled according to population trends. Species whose relative abundances are similar are grouped. Groupings change between regions.
Figure S3: Population estimates for The Chobe District 1993-2012, data obtained from DWNP 2012b and Chase 2011. Species are coupled according to population trends. Species whose relative abundances are similar are grouped. Groupings change between regions.
Figure S4: Population estimates for The Moremi Game Reserve 1993-2012, data obtained from DWNP 2012b and Chase 2011. Species are coupled according to population trends. Species whose relative abundances are similar are grouped. Groupings change between regions.
Figure S5: Population estimates for Ngamiland 1993-2012, data obtained from DWNP 2012b and Chase 2011. Species are coupled according to population trends. Species whose relative abundances are similar are grouped. Groupings change between regions.
S2 Tables

Table S1: Wildlife population estimates between 1996-2012. (Data obtained from DWNP 2012b and Chase 2011)

<table>
<thead>
<tr>
<th></th>
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<th></th>
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<tbody>
<tr>
<td>Elephant</td>
<td>129,687</td>
<td>141,612</td>
<td>146,114</td>
<td>160,316</td>
<td>138,766</td>
<td>168,898</td>
<td>128,340</td>
<td>207,545</td>
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<td>2,993</td>
<td>1,933</td>
<td>909</td>
<td>3,311</td>
<td>34,735</td>
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<td>13,060</td>
<td>11,024</td>
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<td>9,002</td>
<td>10,596</td>
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<td>2,765</td>
<td>3,116</td>
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<td>1,973</td>
<td>3,773</td>
<td>6,054</td>
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<td>67,005</td>
<td>35,628</td>
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<td>38,512</td>
<td>33,900</td>
<td>51,270</td>
<td>114,900</td>
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<td>4,670</td>
<td>6,112</td>
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<td>56,027</td>
<td>41,712</td>
<td>33,246</td>
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<td>9,516</td>
<td>5,729</td>
<td>0</td>
<td>1,875</td>
<td>55,916</td>
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<td>1,033</td>
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<td>2,370</td>
<td>256</td>
<td>415</td>
<td>710</td>
<td>615</td>
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<td>Springbok</td>
<td>4,195</td>
<td>3,164</td>
<td>4,223</td>
<td>2,256</td>
<td>3,986</td>
<td>2,418</td>
<td>1,565</td>
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Table S2: Poaching incidents recorded by the Botswana Department of Wildlife and National Parks (DWNP) 2008-2014.

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<th>2010</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
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<td>1</td>
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<td>0</td>
<td>2</td>
<td>6</td>
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<td>1</td>
<td>17</td>
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<td>18</td>
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<td>4</td>
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<td>0</td>
<td>12</td>
</tr>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Hippo</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
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<td>0</td>
<td>13</td>
<td>13</td>
<td>1</td>
<td>39</td>
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<tr>
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<td>15</td>
<td>25</td>
<td>48</td>
<td>16</td>
<td>2</td>
<td>139</td>
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<td>0</td>
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<td>1</td>
<td>0</td>
<td>36</td>
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<td>0</td>
<td>1</td>
<td>1</td>
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<td>Warthog</td>
<td>5</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>9</td>
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<td>1</td>
<td>0</td>
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<td>0</td>
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<tr>
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<td>0</td>
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<td>11</td>
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<td>14</td>
<td>23</td>
<td>75</td>
</tr>
<tr>
<td>Total</td>
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<td>59</td>
<td>50</td>
<td>86</td>
<td>122</td>
<td>70</td>
<td>67</td>
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</table>
Table S3: Chobe National Park growth rates by Chase 2011. based on linear regression of actual population estimates between 1996-2010 were used to calculate the expected growth of species in 2011 and 2012 and compared against actual population estimates to obtain actual growth rates for 2012.

<table>
<thead>
<tr>
<th>Species</th>
<th>Growth Rate</th>
<th>P value</th>
<th>Expected 2011</th>
<th>Expected 2012</th>
<th>Actual Growth</th>
</tr>
</thead>
<tbody>
<tr>
<td>Buffalo</td>
<td>0.16</td>
<td>0.18</td>
<td>4723.04</td>
<td>932.8</td>
<td>466.4</td>
</tr>
<tr>
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<td>0.28</td>
<td>264.3</td>
<td>23.4</td>
<td>11.7</td>
</tr>
<tr>
<td>Giraffe</td>
<td>0.02</td>
<td>0.41</td>
<td>21.18</td>
<td>10.9</td>
<td>5.45</td>
</tr>
<tr>
<td>Hippo</td>
<td>0.07</td>
<td>0.39</td>
<td>53.9</td>
<td>2.45</td>
<td>1.225</td>
</tr>
<tr>
<td>Impala</td>
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<td>0.18</td>
<td>17.22</td>
<td>161.21</td>
<td>80.605</td>
</tr>
<tr>
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<td>0.83</td>
<td>60.51</td>
<td>1.32</td>
<td>0.66</td>
</tr>
<tr>
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<td>0.03*</td>
<td>30.6</td>
<td>1.9</td>
<td>0.95</td>
</tr>
<tr>
<td>Ostrich</td>
<td>0.04</td>
<td>0.52</td>
<td>16.16</td>
<td>5.12</td>
<td>2.56</td>
</tr>
<tr>
<td>Roan</td>
<td>0.01</td>
<td>0.88</td>
<td>2.23</td>
<td>1.61</td>
<td>0.805</td>
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<td>-2.8</td>
<td>-18.78</td>
<td>-9.39</td>
</tr>
<tr>
<td>Tsessebe</td>
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<td>-2.87</td>
<td>-1.435</td>
</tr>
<tr>
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<td>0.07</td>
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<td>9.1</td>
<td>4.55</td>
</tr>
<tr>
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<td>0.81</td>
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<td>-39.02</td>
<td>-19.51</td>
</tr>
<tr>
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<td>0.54</td>
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<td>17.88</td>
<td>8.94</td>
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</table>
Table S4: Chobe District growth rates by Chase 2011. Based on linear regression of actual population estimates between 1996-2010 were used to calculate the expected growth of species in 2011 and 2012 and compared against actual population estimates to obtain actual growth rates for 2012.

<table>
<thead>
<tr>
<th>Species</th>
<th>Growth Rate</th>
<th>P value</th>
<th>Expected 2011</th>
<th>Expected 2012</th>
<th>Actual Growth 2012</th>
</tr>
</thead>
<tbody>
<tr>
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<td>0.88</td>
<td>74.09</td>
<td>91.11</td>
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<td>0.68</td>
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<td>12.04</td>
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<td>0.69</td>
<td>12.45</td>
<td>10.71</td>
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</tr>
<tr>
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<td>0.25</td>
<td>0.005**</td>
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<td>46.5</td>
<td>-59</td>
</tr>
<tr>
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<td>0.002**</td>
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<td>-1974.5</td>
</tr>
<tr>
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<td>0.4</td>
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</tr>
<tr>
<td>Lechwe</td>
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<td>0.17</td>
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<td>1.52</td>
<td>-192.5</td>
</tr>
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<td>-68.5</td>
</tr>
<tr>
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<td>-38.16</td>
<td>41</td>
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</tr>
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<td>0.68</td>
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<td>166.38</td>
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</table>
Table S5: Moremi Game Reserve (MGR) growth rates by Chase 2011. Based on linear regression of actual population estimates between 1996-2010, growth rates were used to calculate the expected growth of species in 2011 and 2012 and compared against actual population estimates to obtain actual growth rates for 2012.

<table>
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<tr>
<th>Species</th>
<th>Growth Rate</th>
<th>P value</th>
<th>Expected 2011</th>
<th>Expected 2012</th>
<th>Actual Growth 2012</th>
</tr>
</thead>
<tbody>
<tr>
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<td>-20.94</td>
<td>-14</td>
</tr>
<tr>
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Table S6: Ngamiland growth rates by Chase 2011. Based on linear regression of actual population estimates between 1996-2010 were used to calculate the expected growth of species in 2011 and 2012 and compared against actual population estimates to obtain actual growth rates for 2012.

<table>
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<tr>
<th>Species</th>
<th>Growth Rate</th>
<th>P value</th>
<th>Expected 2011</th>
<th>Expected 2013</th>
<th>Actual Growth</th>
</tr>
</thead>
<tbody>
<tr>
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<td>-196.5</td>
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</tr>
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<td>0.11</td>
<td>-571.3</td>
<td>-693.8</td>
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</table>
Chapter 4: Can Mines have a Net Positive Impact on Biodiversity?

A Case Study From Mongolia

Abstract

Mining causes some of the most abrupt and extensive forms of land-use change. It not only impacts biodiversity, but destroys ecological processes and causes land degradation that has cascading effects on biodiversity. Oyu Tolgoi, Mongolia’s largest gold and copper mine, is committed to having a Net Positive Impact (NPI) on biodiversity by the time of mine closure. As a result, the project has implemented a Core Biodiversity Monitoring Program (CBMP) to monitor its impacts on Biodiversity. This program was the first NPI program of its scale and is therefore used as a case study to examine lessons learned and to outline best practices for similar programs that are in their development phase. I propose that focusing mitigation, offsets and monitoring solely on threatened and endangered species and their critical habitat may not be the most effective way to reduce biodiversity impacts. I suggest that restoring and enhancing ecological processes is the most effective mechanism for improving biodiversity; and that careful collaboration with program stakeholders is the key to a successful program.

Introduction

Mining regions experience land-use changes that extend far beyond the footprint of mines, are more abrupt than most other land conversion impacts and expand over time (Sonter et al. 2014). Large-scale ecological processes are often severely degraded because of the extensive water and mineral extraction needs of mining projects. Such extractive industries often pump water from underground aquifers, which can destroy plant life and cause the drying of surface water. Rivers are sometimes diverted to keep open pits from
flooding and leaching can pollute existing waterways. Export roads and railways can bisect critical habitat for many species and the influx of humans to extraction regions can increase poaching and the overharvesting of rare species (Radovani et al. 2015).

In the past decade, Mongolia has received attention for being one of the last places in Asia where large intact habitats and wildlife migrations can still be found. At the same time, it has some of the richest coal and mineral deposits in East Asia. As a result, Mongolia’s expanding extractive industry sector is affecting land-use change across the country, with the majority of mines being situated in the Gobi Desert (Figure 1). The Ömnögovi Province in the South Gobi Desert, where our project is located, has the lowest population density in Mongolia (0.28 people/km²), which was declining in the decade previous to 2010 (Nat. Stats. Office of Mongolia 2010). However, the development of the Oyu Tolgoi (OT) mine reversed this trend, making it one of the fastest growing provinces in the country (2.3% in 2008 — 11.6% in 2010; Nat. Stats. Office of Mongolia 2010).

The Gobi Desert, like most deserts, is highly susceptible to degradation from human impacts. In this fragile landscape, livestock numbers exceed human populations by a factor of 10 (Norton-Griffiths et al. 2014; Nat. Stats. Office of Mongolia 2010) and overgrazing in some areas has already caused shifts in the nutritive value of vegetation (Sasaki 2008). Roads connecting new mines and human settlements have crisscrossed the landscape, leading to severe erosion, especially in and around washes where off-road driving is common. Development is also increasing as a result of the influx of people moving to the area to work for the mine. Despite these new and preexisting cumulative impacts that are degrading rangeland, Mongolia’s largest gold and copper mine, Oyu Tolgoi (OT), is committed to
having a Net Positive Impact (NPI) on biodiversity by the time of mine closure (Oyu Tolgoi ESIA 2012).

This paper explores the results of the first two years of a Core Biodiversity Monitoring Program (CBMP) developed by the Wildlife Conservation Society (WCS) and Sustainability East Asia (SEA), and outlines some lessons learned that may be applicable to other projects that seek to have a net positive impact on biodiversity. While NPI policies represent some of the most cutting-edge strategies for reducing the impacts of land-use change on biodiversity, preliminary data suggests that focusing mitigation, offsets and monitoring solely on threatened and endangered species and their critical habitat may not be the most effective way to reduce biodiversity impacts.

**Background**

The ownership of the OT mine is shared by Rio Tinto and the government of Mongolia. In 2004, Rio Tinto adopted an internal policy requiring that all of its projects result in no net loss of biodiversity (Temple et al. 2012). In addition, loans that the project received from the International Finance Commission (IFC) and the European Bank for Reconstruction and Development (EBRD) triggered environmental regulatory policies requiring that the project demonstrate no net loss (NNL) of biodiversity and a net positive impact (NPI) on biodiversity where critical habitat and endangered species have been identified (IFC 2012; EBRD 2014). In an attempt to achieve NPI, OT has followed the mitigation hierarchy, whereby the project avoids, minimizes, mitigates, rehabilitates/restores and finally offsets its negative impacts on biodiversity (BBOP 2012). The goal is that the gains generated by offsets will be greater than the residual losses of project impacts, allowing the project to demonstrate a net biodiversity gain.
The most endangered species present in the project area is the Asiatic wild ass, known in Mongolia as the khulan (*Equus hemionus*), and it is the principal species for which critical habitat was determined. It is estimated that the global population of khulan is about 55,000, while its population in Mongolia’s Southern Gobi region is estimated to be between 35,000 (95% CI) (Norton-Griffiths et al. 2014) and 39,998 (95% CI = 25,234 – 42,153) individuals (Buuveibaatar & Strindberg 2014; Murphy & Nyamdorj 2015). This represents between 63%-72% of the global khulan population, making khulan the most globally endangered species that will be impacted by the OT project. It is the highest priority biodiversity feature for the project and is listed as Critical in the 2012 Oyu Tolgoi Project Environmental and Social Impact Assessment (ESIA). The khulan range in the South Gobi is estimated to be approximately 94,000 km² (Figure 2), which spans most of the region and consists entirely of rangeland (Buuveibaatar 2014; Murphy & Nyamdorj 2015; Oyu Tolgoi 2012). As a result, OT’s monitoring, mitigation and offset strategies are focused within khulan critical habitat.

OT has two main offset strategies: rangeland improvement and anti-poaching efforts (Murphy & Nyamdorj 2015). Rangeland is a critical habitat required for the survival of khulan and the majority of the other priority biodiversity features. Critical habitat is defined as areas with high biodiversity value, including habitat required for the survival of critically threatened or endangered species and areas having special significance for endemic or restricted-range species (EBRD 2014; IFC 2012). The offset strategy proposes a focus on the landscape comprising the soums (districts) overlapping with the core population of khulan (TBC & IFC 2011). This area contains most, if not all, of the priority biodiversity features, and is where there are significant residual impacts. OT hopes to partially offset its impacts on
khulan and other priority biodiversity features by improving pastureland quality across the entire 94,000 km² khulan range, while also working to ensure that khulan populations are able to cross linear infrastructure and access vital resources such as pasture and water points.

In addition to improved rangeland, OT is addressing illegal hunting, which is the primary threat to the survival of khulan, argali (*Ovis ammon*), goitered gazelle (*Gazella subgutturosa*) and houbara bustards (*Chlamydotis undulate*) (Murphy & Nyamdorj 2015). It is assumed that illegal hunting of wild animals will increase as a result of increased human population drawn to the area by the Oyu Tolgoi project (Radovani et al. 2015). OT plans to offset the indirect poaching impacts associated with the mine’s operations by developing strategies (TBC & FFI 2011) and implementing programs (Oyu Tolgoi 2012) that aim to reduce poaching rates within the khulan range.

The Pilot Core Biodiversity Monitoring Program was therefore focused on collecting baseline data for rangeland quality in the southern Gobi, poaching rates in the region, and the status and impacts on the following priority biodiversity features: khulan, goitered gazelle, argali, short-toed snake-eagle, houbara bustard, Siberian elm trees; tall saxaul (*Haloxylon ammodendron*) forests; and granite outcrop floral communities (Murphy & Nyamdorj 2015). However, we do not discuss argali, saxaul forests or granite outcrop communities in this paper.

**Methods**

This section provides a brief explanation of the methods used to monitor six of the priority biodiversity features, as well as rangeland quality and poaching rates. The section is therefore brief, but a full explanation of methods for each feature can be found in the OT
Core Biodiversity Monitoring Plan (CBMP) through the Wildlife Conservation Society, Sustainability East Asia or The Oyu Tolgoi Project (Murphy & Nyamdorj 2015).

All monitoring protocols are designed to answer the following questions, among others:

1. What are the baseline conditions for each species?
2. How is mining infrastructure impacting species movements, breeding success, and avoidance behavior?
3. Is mining infrastructure fragmenting critical habitat?

**Ground-based ungulate surveys.** Asiatic wild ass, goitered gazelle, argali, and Mongolian gazelle were all monitored using ground-based distance sampling methods (Thomas et al. 2010). A systematic survey design with a random start was generated using the Distance 6 software. The survey design consisted of 29 transects with spacing of 20 km totaling 4,820 km of survey effort. Owing to the ruggedness of the topography (mountains and sand dunes), 28% of the total transect length was truncated, which resulted in 64 transects (range = 4 – 205 km) with a total of 3,464 km of survey effort (Buuveibaatar & Strindberg 2014; Murphy & Nyamdorj 2015).

Transects were driven in the spring and fall of 2013 and in the fall of 2014. B. Buuveibaatar of the Wildlife Conservation Society led surveys and data analysis. The location of other environmental factors that might affect animal density were also recorded or gathered and include: a Normalized Difference Vegetation Index (NDVI), vegetation, altitude, surface water, roads and human settlements. Analysis was conducted using Distance software and generalized linear models. Outputs include: 1) Estimated population size (with confidence limits) in the survey area for Asiatic wild ass and goitered gazelle; 2) mapped
species distributions; 3) population size and distribution related to the distance from the road and other natural and human factors in each 5 x 5 km survey block; and 4) flight distances (mean, min., max. and variance) for each species from vehicles (Buuveibaatar & Strindberg 2014; Murphy & Nyamdorj 2015).

**Aerial ungulate surveys.** The aerial surveys recorded population densities for Asiatic wild ass, goitered gazelle, argali, Mongolian gazelle and livestock. A grid totaling 30,000 km of transect was flown. Photographs and thermal images were taken during flights (Norton-Griffiths et al. 2014).

M. Norton-Griffiths and H. Frederick conducted the aerial survey once in 2014. Multiple parties conducted data analysis. Data on other environmental factors that might affect animal density were also gathered and include: NDVI, vegetation, altitude, surface water, roads and human settlements. Analysis of survey data included: 1) the counting of individual animals using photographs from the aerial survey; 2) estimating population size (with confidence limits) in the survey area for Asiatic wild ass and goitered gazelle; 3) mapping species distributions; 4) assessing population size and distribution related to the distance from the road and other natural and human factors at a resolution of 5 x 5 km; 5) and an assessment of road density (Norton-Griffiths et al. 2014).

**Collared Asiatic wild ass and goitered gazelles.** Twenty Asiatic wild asses were collared (13 stallions and 7 mares) with satellite transmitting GPS locators. A total of 10 goitered gazelles (5 female and 5 male) were also collared in Khatanbulag soum and the Umnugobi and Dornogobi Provinces. However, two animals died and 8 remained collared (Kaczensky & Payne 2014; Buuveibaatar 2014; Murphy & Nyamdorj 2015).
Using satellite telemetry, our goal was to determine the following: 1) how much khulan and gazelle habitat is indirectly lost due to both species avoiding roads, power lines, houses, livestock and other human developments; 2) which factors affect khulan distribution and the ways in which their habitat selection determines their response to human disturbance. Sub-goals include: 1) to what extent is the distribution of ungulates negatively correlated with the distribution of livestock; 2) to what extent do ungulates avoid houses and towns; 3) to what extent does ungulate avoidance behavior result in fragmentation of habitat; 4) how dependent are species on fixed water sources; and 5) how do ungulates respond to environmental forces such as variation in precipitation and temperature and resultant changes in vegetation (Kaczensky & Payne 2014; Buuveibaatar 2014; Murphy & Nyamdorj 2015)?

Khulan collars were fixed in August of 2013 and have an automatic release mechanism that allows them to drop off after 2 years. Collars that did not drop off after 2 years were removed in September of 2015. Individuals where then refitted with 20 new collars (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). Goitered gazelle collars were fixed between September 20th and January 25th, 2014. Any collars that do not fall off will be removed in the fall of 2016 and new ones will be refitted (Buuveibaatar 2014).

Kaczensty and Payne were responsible for khulan collaring and data analysis. Buuveibaatar from WCS was responsible for gazelle collaring and data analysis. Additional data on environmental factors that might affect animal densities was also gathered and used in analysis. This included: NDVI, vegetation, altitude, surface water, roads and settlements. Analysis of collaring data included: recording all location and activity signals; mapping individual movements; mapping distributions related to the distance from roads and other natural and human factors (such as water points); number, frequency and location of points.
where animals cross infrastructure (notably surfaced roads but also including major un-
surfaced roads, power lines and any railways) or appear to turn back from or walk parallel to
infrastructure; location and cause of mortality, including annual losses due to hunting
(Kaczensky & Payne 2015; Murphy & Nyamdorj 2015).

**Ungulate carcass monitoring.** Using two vehicles, seven 40x40 km grids were
searched for Asiatic wild ass, goitered gazelle, argali, and Mongolian gazelle carcasses.
Transects were driven once in the fall of 2014 and 2015 and coincide with peaks in recorded
poaching incidents. When carcasses were located, cause of mortality, age and sex of each
individual were determined using methods detailed in Batsaikan, 2015 and Murphy and
Nyamdorj, 2015.

Individuals from the Mongolian National University conducted surveys on an annual
basis and data were analyzed by Strindberg and Buuveibaatar (2014). Additional data was
also collected on NDVI, vegetation, altitude, and the proximity of carcasses to surface water,
roads and settlements. Analyses were conducted to determine mortality and poaching rates.
Outputs include: mapped locations of carcasses for Asiatic wild ass and goitered gazelle;
number and location of carcasses related to distance from roads and other natural and human
factors; and standardized mortality rate (no. carcasses / survey grid) and poaching rate (%
mortalities caused by hunting) (Batsaikan 2015; Strindberg & Buuveibaatar 2014; Murphy &
Nyamdorj 2015).

**Houbara bustard monitoring.** Transects were driven perpendicular to linear
infrastructure. The field team stopped every 1 km to look for bustards and recorded type of
activity and number of individuals when birds were encountered (Purev-Ochir et al. 2015;
Murphy & Nyamdorj 2015).
The Wildlife Science and Conservation Center (WSCC) conducted surveys each spring on an annual basis and was responsible for all data analysis. Data was also collected on: vegetation near bird sighting, altitude, roads and human settlements. Data analysis included: comparing the number of individuals to their distance from infrastructure; plotting the number of individuals related to distance from roads; number of bustards recorded within each km² (Purev-Ochir et al. 2015; Murphy & Nyamdorj 2015).

**Short-toed snake-eagle monitoring.** Each spring, all elm trees and other potential nesting sites within a 20-kilometer radius of the OT mine site were searched for short-toed snake-eagle nests. Nests were also inspected and all nesting activity recorded, including number of eggs and hatchlings.

Surveys were conducted in the early spring on an annual basis by the WSCC (Gungaa et al. 2014; Murphy & Nyamdorj 2015).

Data were also collected on vegetation within close proximity to nests sites, altitude, roads, human settlements and nest tree characteristics. Data analyses included: mapping the distribution of nests in relation to natural and manmade features (Gungaa et al. 2014; Murphy & Nyamdorj 2015).

**Elm, saxaul and understory vegetation monitoring.** Elm trees, tall saxaul forests and their understory vegetation were monitored within the Gunii Hooloi catchment. The Gunii Hooloi is a large underground aquifer from which the OT mine obtains most of its water. Two control sites (one for elm and one for saxaul) were selected in 2014 and placed outside of the areas impacted by groundwater drawdowns in the Gunii Holoi Catchment and Undai River. Six sites were randomly chosen for elm and six sites randomly chosen for saxaul using an ArcGIS random selection tool. Within each site 12 individual trees were
sampled. Within each of the 12 sites, the following data was recorded: composition and structure of understory vegetation; forest age structure and species composition; and general tree health indicators, which included: percentage of dead material on each tree, pest infestations and composition and structure of forests (Murphy 2014; Murphy & Nyamdorj 2015).

Monitoring was conducted on a biannual basis, in the spring and fall of each year (2013-2015) by the WCS vegetation monitoring team. Data analysis included: using Generalized Linear Mixed Models (GLMMs) to investigate the relationship between variables associated with elm and saxaul tree size and variables such as percent of the tree categorized as dead, the percent leaves eaten by insects (only for elm trees), soil type, slope and elevation. The following variables were also recorded: recruitment rate of trees, age structure; and percentage of bare ground within sites (Murphy 2014; Murphy & Nyamdorj 2015).

**Rangeland health.** To assess rangeland health, the Ecological Site Description (ESD) concept was applied (NRCS 2016) to 32 OT vegetation monitoring plots in Khanbogd soum and State Transition Models (STMs) were developed for the most common rangeland communities (Stringham et al. 2003). This model defines the ecological potential of the plant community and indicates what successional state the rangeland community is in (Ankhtsetseg 2014; Murphy & Nyamdorj 2015).

Monitoring included collecting data on species structure and composition, soil characteristics, and landforms associated with the monitoring site. The OT Environment Team conducted monitoring on an annual basis and data were analyzed by the GreenGold Project (Ankhtsetseg 2014; Murphy & Nyamdorj 2015). Data were analyzed according to the
ESD protocols developed by the NRCS and sites were classified according to State and Transition Models (Stringham et al. 2003).

Results

Khulan collaring. Analysis of collared khulan movements from Oct 2013 - Dec 2014 show individual ranges of 9,934 – 63,431 km. and a total range of 94,388 km. (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). Movement patterns also showed that 22% of all recorded khulan were aggregated in one location along a fence line (location not given to protect the species from poachers) comprising 6% of the total khulan range (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). Similarly, 29% of all Khulan locations were recorded in two other locations, which comprises 17% of the area; both areas are important seasonal habitats (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). In addition, several specific water points have been identified as key features for khulan range use. While analysis is still underway, preliminary analyses indicate that specific locations are important breeding range for khulan (Kaczensky & Payne 2015; Murphy & Nyamdorj 2015).

The OT ESIA (2012) identified potential impacts on khulan due to the avoidance of the mine site and access roads. However, 12 of the 20 (60%) satellite-tracked animals came within the vicinity of the OT road or mine (within ≤10 km). Of the twenty khulan collared and tracked, 11 crossed the OT road 100 times and seven khulan crossed the coal export road 34 times, demonstrating that these roads are not an absolute barrier. (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). However, a comparison between expected and observed crossing frequency suggests that the OT road between the mine and the Mongolian-Chinese border crossing point may reduce crossings to only 41% of what would otherwise be expected based on khulan presence in the vicinity (Kaczensky & Payne 2014; Murphy &
Furthermore, the timing of khulan crossings is largely restricted to periods of low traffic at night (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015).

**Goitered gazelle collaring.** Baseline data for goitered gazelle estimates the Southern Gobi population to be approximately 32,614 (95% CI = 25,234 – 42,153) (Norton-Griffiths et al. 2014; Buuveibaatar & Strindberg 2014; Murphy & Nyamdorj 2015). Indicators associated with habitat loss due to avoidance of infrastructure show that, of 8 collared gazelle, the average cumulative distance travelled during the four-month survey period reached more than 3,500 km (Buuveibaatar 2014; Murphy & Nyamdorj 2015). For all tracked animals, the average home range size was 1,224. The mean home range size of the collared gazelles near the OT mining site was substantially smaller than those collared at the control sites in Khatanbulag soum (Buuveibaatar 2014; Murphy & Nyamdorj 2015), which is a significant distance from any mining activity ($t = 2.82$, $p = 0.03$). Of the animals collared near the OT site, one female gazelle crossed the OT road 37 times and crossed the nearby coal export road only twice. The male did not cross either road (Buuveibaatar 2014; Murphy & Nyamdorj 2015). Movement trajectories of the collared gazelles show that they became dependent on surface water at the end of November 2014, probably due to lack of snow cover in 2014 (Buuveibaatar 2014; Murphy & Nyamdorj 2015).

**Aerial survey.** The majority of results from the aerial survey are provided under the species headings, as estimating population sizes was the main goal of the survey. However, additional data relevant to OT’s impacts on biodiversity were also obtained.

The aerial survey report (Norton-Griffiths et al. 2014) indicates that there is a disassociation between wildlife and livestock ($Pearson\ Chi-square = 0.025$, $df = 1$; Pearson correlation coefficient = -0.005, $df = 4,233$), indicating that wildlife avoids livestock...
Photographs that represent the scale of observation in this case have an average area of 0.023 km² and almost no associations were detected. The OT aerial survey report indicates that if wildlife and livestock were randomly associated then “there would be some 130 grid cells in which both were present,” whereas only 17 were observed (Norton-Griffiths et al. 2014).

Out of 51,349 JPEG images captured along the flight lines, only 4,271 (8.3%) contained animals or human impacts. Of those, 474 had ungulates totaling 1,161 individuals, 409 had livestock totaling 11,945 individuals and 3,388 showed indications of human impacts, such as settlements, cultural sites, roads or livestock structures (Norton-Griffiths et al. 2014). Wildlife also exhibited a strong negative relationship with human impacts (t = -3.270, p = 0.001) and a statistically insignificant negative relationship with livestock (Norton-Griffiths et al. 2014) (t = 0.523, p = 0.601).

The aerial survey team also isolated the road index from other human impacts, which produced similar results (t = -2.170, p = 0.030), indicating a strong road avoidance behavior. A heat map of road density was also generated by the aerial survey team, with a 10 km kernel radius (Norton-Griffiths et al. 2014). The map is based on road density estimates from the full set of road counts, (39,138 photographs) and indicates that approximately 30% of the surveyed area has been impacted by roads. It also highlights the area between the OT transport road and the energy resources road as a heavy impact area (Norton-Griffiths et al. 2014).

Carcass surveys. Khulan carcass surveys suggest that poaching rates (based on recent Khulan carcasses < 2 years old) initially increased during the 2004-2006 period, but have declined in 2013/2014 (Batsaikan 2014; Strindberg & Buuveibaatar 2014; Murphy &
Poaching rates seem to be somewhat biased towards female animals (Strindberg & Buuveibaatar 2014). Comparisons of khulan carcass and live population densities for 2014 indicated that 3% (about 1,204 khulan) of the population may be impacted by poaching (Strindberg & Buuveibaatar 2014). Projected estimates for gazelle poaching are based on the proportion of four goitered gazelle carcasses observed for every one khulan observed by the mobile anti-poaching unit, indicating that likely well over of 15% (about 3,785 Goitered gazelle) of the gazelle population is impacted by poaching (Strindberg & Buuveibaatar 2014).

**Short-toed snake-eagle and Hubara bustard.** Baseline data for snake-eagle and Houbara bustard populations are vague due to their low population densities. Only four nesting pairs of snake-eagles were observed in 2013 and seven in 2014 (Gungaa et al. 2014; Murphy & Nyamdorj 2015). Houbara bustard density was estimated at 0.0008 ind/km (Batbayar 2014; Murphy & Nyamdorj 2015). In 2013, the field team observed only one snake-eagle chick per nest and three of four fledged (Gungaa et al. 2014; Murphy & Nyamdorj 2015). In 2014, the field team also observed one chick per nest and five of seven fledged (Gungaa et al. 2014; Murphy & Nyamdorj 2015). However, the field team found 10 other species of raptors in 2013 and 352 nests (Gungaa et al. 2014; Murphy & Nyamdorj 2015). In 2014, they found 11 species of raptors and 152 nests (Gungaa et al. 2014; Murphy & Nyamdorj 2015).

**Elm and saxaul tree and understory health.** The mean percentage of dead canopy on elm trees fluctuated between 37% and 10% (Figure 3) between 2013 and 2014, respectively. Species diversity in the understory was between 12 and 13 species for elm (Murphy 2014). The percentage of understory cover for elms was 11% in 2013 and 15% in
2014 (Murphy 2014). Between the two years, the number of understory height classes remained static at four species (Murphy 2014; Murphy & Nyamdorj 2015). Bare ground exposure was higher than 75% for both elm and saxaul sites in both years (Murphy 2014; Murphy & Nyamdorj 2015). Eight pest species were identified at elm monitoring sites, however pests were impacting less than 10 % of foliage (Figure 4). The average diameter at breast height (DBH) for all 75 elm trees was 65 cm, indicating that most trees are 100-150 years old (Figure 5). In addition, zero recruitment of elm seedlings in 90% of monitoring sites was recorded (Murphy 2014; Murphy & Nyamdorj 2015).

**Rangeland health.** Of the 32 rangeland monitoring plots, 80% were in the reference state, indicating a potential for rangeland recovery from grazing impacts. Rangeland sites consisted of an average of eight species and the average vegetation cover at each site was 20% (Ankhtsetseg 2014).

**Discussion**

Whereas the first two years of core biodiversity monitoring produced the desired results and addressed many of the questions laid out by the OT ESIA (2012) and Offsets Strategy (2011), it became clear that the priority biodiversity features outlined previous to the development of the core biodiversity monitoring did not fully address the potential impacts the OT project could have on biodiversity. It also does not monitor OT’s impacts on abiotic factors such as soil erosion and water availability, which govern rangeland health and which may be having a bigger impact on individual species than infrastructure avoidance.

Similarly, some of the impacts on biodiversity are a result of cumulative impacts from other mines and national infrastructure projects, such as railways and border fences. These cumulative impacts require multiparty collaboration and sharing of data. Conflicting private
sector, national and international offset and mitigation policies also create roadblocks for NPI projects that can be avoided with focused collaboration efforts.

**Erosion.** Studies conducted by Murphy (2014) indicated an average of 77% bare ground along 15 transects (Table 1). These data are supported by similar findings by the GreenGold Project (Ankhtsetseg 2014; Murphy & Nyamdu 2015). Murphy (2014), Ankhtsetseg (2014) and Sasaki et al. (2008) all found that the majority of their sample sites in the Gobi were characterized by sandy soils with high erodibility.
Photos from the aerial survey conducted within the 94,000 km² khulan range were analyzed by the OT Aerial Survey Team and it was also shown that over 30% of the landscape has been impacted by roads (Norton-Griffiths et al. 2014), the majority of those being unpaved dirt tracks (Image 1). Roads, and especially dirt roads, created by vehicles repeatedly driving in the same place, have been shown to significantly degrade rangeland habitat through erosion (Zeedyk & Clothier 2014). Roads can cause gullying and the channelization of alluvial fans and shallow desert washes if not properly graded (Image 2). Channelization and gullying lowers groundwater tables.

Roads also drain water off landscapes, leaving them dry, and deposit water into landscapes that are not capable of absorbing it, leading to additional erosion (Zeedyk & Clothier 2014). Water that travels along roads also moves much faster than it normally would and causes severe erosion where it does finally reconnect with landscapes (Image 3).
This does not only occur on dirt tracks, but is even more pronounced along paved roads that are improperly graded and drained. Culverts and flood mitigation structures along the OT transport road are having clear negative impacts on the surrounding landscape by causing erosion where water outlets have been placed (Image 5). This type of erosion creates head-cuts that channelize water even more; channelized water moves faster than water that is spread out (Image 3), incising channels and lowering water tables (Zeedyk & Clothier 2014).

**Water.** In the Gobi Desert, water scarcity is perceived as the most critical threat to both herders and wildlife (Oyu Tolgoi 2012; Murphy & Nyamdorj 2015). Water has been identified by all of the OT core biodiversity monitoring teams as a key resource influencing species abundance, resilience and movements (Murphy & Nyamdorj 2015). As a result, each species report within the larger OT Core Biodiversity Monitoring Report suggests that OT mitigation efforts focus on the protection of critical water sources. Herders blaming OT for groundwater reductions have been documented on many occasions, both in the ESIA (Oyu Tolgoi 2012) stakeholder engagement process and in the local media (Tolson 2012). General sentiments of perceived water scarcity (74% of surveyed community members believe there are not enough wells in the region) support the notion that groundwater levels have decreased (Oyu Tolgoi 2012).
The OT mine extracts water needed for mine operations from the Gunii Hooloi Aquifer (Oyu Tolgoi 2012). Detailed analysis conducted by Aqua Terra and other local hydrological monitoring firms has shown that there is almost no connection between shallow and deep aquifers in the impacted area (Oyu Tolgi 2012). It is, therefore, unlikely that shallow groundwater reductions are due to OT’s use of water from the deep aquifer. Shallow water table reductions are most likely due to the type of localized erosion mentioned above that is channelizing water and lowering water tables. This can be attributed to overgrazing, off road driving and improper drainage of paved roads (Zeedyk & Clothier 2014).
Shallow water tables are particularly important in the Gobi Desert. Local people hand-dig wells in washes to provide water for livestock and their household needs (Oyu Tolgi 2012). Khulan and other ungulates dig for shallow groundwater (Image 4), which is particularly important for their survival (Kaczensky et al. 2005). Other species opportunistically use water pits that have been excavated by khulan (Kaczensky et al. 2005). Similarly, birds and many other species rely on desert seeps that dry up when groundwater tables drop, and the productivity of rangeland vegetation depends on its connection to shallow water tables.

Image 4: Left, waterholes in washes dug by khulan; right, khulan digging water hole and drinking (photos by Kaczensky).

Khulan. Khulan movement patterns suggest that khulan repeatedly return to specific water points and that those water sources are critical for their survival (Kaczensky & Payne 2014). Some of these are areas where shallow groundwater can be accessed through digging and some are surface waterholes. The khulan collaring data also provides hot-spots where important khulan habitat can be found (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). One hot-spot exists along a portion of the 95 km OT transport road (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015). This area of the road provides high quality
rangeland for khulan and also consists of several waterholes (Kaczensky & Payne 2014; Murphy & Nyamdorj 2015).

Severe erosion can be observed along this road (Image 5). Currently, the impacts of erosion on rangeland quality have not been verified, but if conditions persist, it is likely that decreases in ground water levels will begin to degrade this important khulan habitat. In addition, surface water levels (water holes) are reduced when groundwater levels are lowered. Khulan are also unable to dig for water where shallow water tables have been lost (Kaczensky et al. 2005).

![Image 5: Left, constriction of the OT Transport rd. Right, erosion beginning only a few months after construction was completed](image)

Roads were also shown to cause avoidance behavior in khulan. The data clearly suggest that khulan do avoid roads to some degree. However, the large number of crossings recorded is most likely due to the fact that traffic on the road is currently low. The road was completed in 2014 and it is likely that traffic will consistently increase as Mongolians learn that driving to China is feasible given the newly paved road connecting the two countries.
However, khulan habitat does not currently appear to be severely fragmented by the road. Fencing along railway lines and along the Mongolian Chinese border, on the other hand, has been shown to be a complete barrier (Figure 6) to khulan and other ungulate movements (Kaczensky & Payne 2014). WCS and the OT project have already addressed this issue.

The Mongolian Ministry of Environment and Green Development and the Ministry of Transportation participated in a study tour, hosted by WCS, on the impacts of linear infrastructure on wildlife, along with several workshops that compelled them to create a Joint Ministerial Working Group on the subject. The joint working group subsequently created new regulations aimed at minimizing the impacts of roads and railways on wildlife movements. The first goal of the joint ministerial working group was to remove the fence along the Trans-Mongolian Railway line. OT funded the removal of the fence, which compelled lenders to include the effort as an offset gain for the mine. This provides a perfect example of how collaboration between the private sector, local conservation organizations, national governments and international lenders can solve key issues impacting critical habitat and even change regulatory policies.

**Goitered Gazelle.** The issues listed for khulan are the same for goitered gazelle. While it is hypothesized that goitered gazelle gain a large portion of their water from forage, analysis of their movement patterns also suggest that they rely on surface water and shallow groundwater during dry periods (Buuveibaatar 2014; Murphy & Nyamdorj 2015). Fences also restrict their movements. The removal of the Trans-Mongolian Railway fence connected two previously fragmented herds and increased the potential range of this species (Buuveibaatar 2014; Murphy & Nyamdorj 2015).
Elms and rangeland quality. Indicators for tree health and rangeland quality are much more complex and, from the beginning, should have been designed to determine if ground water drawdowns in the Gunii Hooloi aquifer and diversions in the Undai River would impact tree and rangeland health. While it was noted that tree and other monitoring sites needed to be coupled with piezometers during pre-monitoring planning, several variables hindered the coupling of these data sets (Murphy 2014; Murphy & Nyamdorj 2015). Assumptions made during the ESIA processes and the setting of priority biodiversity features made it difficult to advocate for the importance of this type of monitoring. The focus on endangered species as a priority and the assumption that tree health monitoring could be done adequately without an understanding of groundwater dynamics made funding for groundwater monitoring difficult to acquire in the first year.

Agreements were later made to use preexisting piezometers installed by the OT project to measure groundwater fluctuations. Additional piezometers were also installed by the OT project in order to assist with tree and understory vegetation monitoring. However, obtaining this data after field seasons for analysis became difficult. The OT mine is a huge industrial operation. The shifting of personnel within the company, strict rules about data sharing and lack of communication between the biodiversity and water departments within the company made it difficult to obtain the necessary data in time for quarterly reports. As a result, data on groundwater fluctuation are not available here.

Additional methodologies, such as the placement of dendrometers on trees, are also needed to decouple rainwater and groundwater impacts on tree health. This will assist in detecting changes to elm health before tree mortality in this region is imminent. Similarly,
exclosures are needed to assist in decoupling the effects of grazing from both precipitation and groundwater drawdowns on elm seedling recruitment and NPP.

Elm surveys (Murphy 2014; Murphy & Nyamdorj 2015) indicated no regeneration of elm trees within riparian areas and showed evidence of small diameter trees being eaten to a hedge-like state by ungulates (Liu et al. 2013; Murphy 2014; Murphy & Nyamdorj 2015). Elm seedling recruitment relies on shallow ground water tables that keep soils moist long enough to recruit new trees. In addition, most elm trees in the South Gobi are between 150 and 200 years old according to ring samples taken by OT (2012) and diameter measurements (Table 2) taken by Murphy (2014). These findings are supported by Wesche et al. 2011. The average lifespan of elm trees in their native habitat is 100-150 years, with an average trunk size of 66 cm (Townsend 1975). Thus, most elm trees in the project area have either reached or exceeded their lifespan. When old trees begin to die, lack of seedling recruitment may push the South Gobi into a state where elm trees do not exist and recruitment is impossible without human interventions such as restoration of water tables, reseeding, and the removal of herbivores or the use of enclosures.

While local studies on grazing pressure in the Ömnögovi Province suggest moderate to low levels of grazing, loss of elm tree recruitment and documented signs of hedging (Murphy 2014; Murphy & Nyamdorj 2015; Wesche et al. 2011; Lui et al. 2013) caused by browsing ungulates suggest the opposite. Low grazing estimates may be due to lack of reference data resulting from the slow increase in grazing pressure in this region over several hundred years (von Wehrden et al. 2006; Wesche et al. 2011). Studies by von Wehrden et al. (2006) indicating that anthropogenic pressures have reduced elm distribution in the South Gobi over the past 100 years support this hypothesis. The fact that the majority of elms in the
study area were at or beyond the age of mortality suggests that overgrazing has indeed
persisted for at least 100 years (Murphy 2014; Murphy & Nyamdorj 2015). The lowering of
groundwater tables is likely affecting older trees as well as seedling recruitment and may lead
to the complete loss of the species in the region within the next decade.

Perhaps most importantly, grazing gradient studies by Sasaki et al. (2008) have
shown that grazing pressure is highest around water sources and that such intense levels of
grazing have created threshold shifts that have replaced nutrient-rich grasses and forbs with
nutrient-poor annual weedy forbs and shrubs. While grazing in Mongolia has persisted for
over 5000 years, the increase in cashmere goats over the last 100 years (Romanova 2012) has
increased the number of animals grazed by 300% as well as shifted the composition of
livestock (from camels and horses to goats and sheep) in the Gobi (Romanova 2012; Porter et
al. 2016).

There is increasing evidence that the Ömnögovi Province of the southern Gobi is
degraded (Murphy & Nyamdorj 2015; Wesche et al. 2011; Liu et al. 2013). Signs of
degradation verified in the Southwestern United States, such as shrub encroachment and
severe erosion, are similar to rangeland conditions in some areas in the Gobi, which supports
this hypothesis. In addition, some features seen on the landscape such as very shallow hand-
dug wells, deflocculated clay soils, salt accumulation, relict organic soil layers and hardpan
are all indications that some landscapes in the Gobi may have previously been wet meadows
or even wetland environments.

**Birds.** Available data on bird nesting sites from 2013-2015 suggests that the
Khanbogd area is globally important for many small falcons and other raptors (Gungaa et al.
2014; Murphy & Nyamdorj 2015). It has been suggested that nest sites and the density of
birds of prey documented to date in this area may be some of the highest for nesting raptors in Asia (Gungaa et al. 2014; Murphy & Nyamdorj 2015). Simply recording baseline data and nesting success rates for snake-eagles would not help us understand the connection between elms trees as important nesting sites and the potential decline of raptors in the region. It therefore may be wiser to monitor whole communities (such as raptors or rodents) instead of simply monitoring the nesting success rate and avoidance behavior of a few threatened and endangered bird species.

**Anti-poaching offset.** The OT ESIA (2012) identified two offset needs. The first is an anti-poaching program designed to reduce poaching in the offset landscape. The second is the implementation of rangeland improvements. These programs are not technically part of the CBMP, but are put out to bid as separate contracts. The WCS/SEA Team submitted a proposal to manage the anti-poaching offset and won the contract.

The WCS/SEA Team proposed a comprehensive anti-poaching initiative that had been successfully tested in Northern Mongolia and in many other countries where WCS operates, including other parts of Asia (Lynam, 2004; Lynam, 2005). Rationale for the proposed methodology suggest that poaching can be reduced if border guards and rangers have the legal mandate to enforce environmental laws (Badam 2006), if they themselves respect the law, if staff capacity can be raised to enable environmental law enforcement (Heffernan et al. 2005; Lynam, 2006), and if they are able to effectively coordinate wildlife enforcement activities with other relevant agencies.

One of the main issues anti-poaching patrols face is an inability to prosecute offenders due to the fact that those involved in the patrol lack the authority to arrest and/or prosecute offenders. It is therefore necessary to create a Multi-Agency Team (MAT) whose
authority is multi-jurisdictional and can prosecute border, protected area, and local level poaching incidents. The MAT is responsible for making sure that poaching cases are recorded, tried and that those convicted are prosecuted appropriately. They also provide oversight and support for Mobile Anti-Poaching Units (MAPU) that were also formed by the WCS/SEA Team. MAPUs report directly to the MAT, who keep records on all poaching incidents and handle enforcement and convictions.

This system provides agencies who would otherwise not have the authority to arrest poachers, and hence no incentive to chase or stop poachers, with the authority to do so. The MATs consist of representatives from the following agencies in Mongolia: 1) the Environmental Protection Agency (EPA); 2) the Protected Area Administration (PAA); 3) the aimag-level (local level) branch of the national State Police; 4) the Specialized Inspection Agency (SIA); 5) the Intelligence Agency (IA); 6) Customs (borders only); and 7) the General Justice Agency.

The Mobile Anti-Poaching Units (MAPUs) are responsible for conducting anti-poaching patrols, reaching out to inform and engage community members and reporting to the MAT. The WCS/SEA Team facilitated the creation of three Mobile Anti-Poaching Units (MAPUs) and provided initial training in team building, laws and legislation, how to conduct patrols and searches, and the use of reporting and data collection tools among other things. Individuals from the multiple agencies above also comprise the MAPUs, but community members are included in patrols as well.

The Spatial Monitoring and Reporting Tool (SMART) was then introduced to MAT and MAPU Teams to improve anti-poaching efforts and overall law enforcement effectiveness by providing a standardized tool to collect, store and evaluate data on patrol
efforts (e.g., time spent on patrols, areas visited, and distances covered), patrol results (e.g.,
snares removed, arrests made) and threat levels. When used effectively, SMART can create
and sustain information flow between ranger teams, analysts and conservation managers, and
has been shown to significantly reduce poaching in 27 countries worldwide.

In addition, the anti-poaching initiative has created a secretive community informant
network with a hotline designed to catch poachers that MAPUs are unable to detect.

This initiative has just completed its first year in operation, but results from similar
initiatives in Mongolia and around the world suggest that it will be very successful (Lynam,
2004; Lynam, 2005). It represents the best program developed using CBMP results.

**Rangeland offset.** On the other hand, the rangeland offset strategy has not been
developed for the OT project and is without a doubt the most complex. It represents the most
important strategy for increasing biodiversity in the Gobi. It encompasses habitat for not only
all of the critical biodiversity features listed in the ESIA (2012) but is also important for the
survival of all biodiversity in the South Gobi. I propose that rangeland improvements can be
made simple by employing methods that have been refined over the past decade in the
Southwestern USA.

Changes in rainfall regimes and temperature increases are predicted to impact NPP
and plant community composition across the globe (Seager et al. 2007; Gutzler & Robbins
2011). Plant productivity, water stress and soil biochemistry are strongly governed by soil
moisture dynamics that are often unpredictable (Knapp et al. 2002; Austin et al. 2004). As
the impacts of climate change increase, it is hypothesized that in arid ecosystems, shifts in
rainfall events will be characterized by less frequent and more severe storms that produce
more precipitation at one time (Seager et al. 2007; Gutzler & Robbins 2011). Shifts in
rainfall regimes that produce fewer storms of higher intensity have been shown to reduce NPP in grasslands due to reduced infiltration rates and reduced plant nutrient uptake (Cai et al. 2014; Collins et al. 2008; Knapp et al. 2002; Austin et al. 2004; Petrie et al. 2015; Knapp et al. 2015).

Shifts in rainfall modify the temporal patterns of plant water stress, which impairs the ability of plants to assimilate nutrients, water and carbon (Collins et al. 2008; Petrie et al. 2015; Austin et al. 2001; Knapp et al. 2002). In addition, the increased intensity of storms reduces soil infiltration rates causing flash floods and erosion that lowers the elevation of stream channels, which leads to the lowering of water tables (Valentin et al. 2005; Rosgen 1996; Ffolliott, & DeBano 2005).

In the Southwestern United States, it is widely accepted that anthropogenic forcings such as roads, railways (Ffolliott, & DeBano 2005), logging and overgrazing (Kauffman & Krueger 1984; Fleischner 1994; Trimble & Mendel 1995) have caused the degradation of 50% of local riparian areas, wetlands and alluvial fans (Jenson & Platts 1990, Tausch et al. 2004). In arid grasslands, climate change and grazing have been shown to facilitate shrub encroachment (Caracciolo et al. 2016; Van Auken 2000; Collins & Xia 2015). Shrubs reduce groundcover (Baez & Collins 2008), which degrades soil characteristics and increases its erodability (D’Odorico et al. 2012; Van Auken 2000).

Erosion leads to stream, wetland and alluvial fan channelization that lowers water tables and reduces soil moisture content (Valentin et al. 2005; Hammersmark 2008, 2009a, 2009b). At this point, reduction in grazing intensity and reseeding will not restore native grasses (Suding et al. 2004; van de Koppel et al. 1997; Van Auken 2000). This is because meadow and riparian grasses, forbs, and facultative wetland species are dependent on their
connection to shallow groundwater tables as well as the ability of surface water to infiltrate soils (Loheide et al. 2009; Peitre et al. 2015).

Dense groundcover and grasses slow runoff from storm events and allow water to infiltrate soils. Ladwig et al. (2015) have shown that in the desert environment, hydrolytic enzymes are higher under plants than in the “unvegetated interspace”. These enzymes stimulate microbes that help make nutrients more available to plants through fungal transfers (Collins et al. 2008; Austin et al. 2004; Peitre et al. 2015). This increases the nutrient uptake of plants and improves photosynthesis. Without groundcover, such as forbs and grasses, soil loss increases and water tables are lowered further (Valentin et al. 2005).

When shallow groundwater tables are lost, grasses and forbs are replaced by additional woody and invasive species with deeper roots (D’Odorico et al. 2012; Valentin et al. 2005; Van Auken 2000). Total ground cover is again reduced and seed banks are polluted with invasive species that outcompete native vegetation under new moisture regimes. In this way, positive feedback loops are created that perpetuate alternative stable states (Suding et al. 2004). Similarly, overgrazing can reduce riparian vegetation, causing erosion. This leads to the additional channelization of drainage patterns, which leads to a further reduction, and in some cases total loss, of vegetation (Fleischner 1994; Trimble & Mendel 1995).

As was seen in previous sections of this case study, erosion from roads and other forms of land-use change have had a similar effect on rangeland (Ffolliott, & DeBano 2005). Thus, this single anthropogenic forcing (land conversion) pushes both rangeland and riparian areas toward ecological state shifts that are difficult to reverse. Restoration often requires the forceful disruption of feedback loops (Suding 2004; Suding & Hobbs 2009), such as the manipulation of abiotic processes and the assisted reestablishment of native species, as well
as the removal of the perturbation that caused the shift (in this case, erosion) (Suding & Hobbs 2009). Rangeland ecologists are increasingly using alternative state models that incorporate these feedback loops and internally reinforced states as indicators of potential system collapse (Suding 2004; Schroder et al. 2005; Scheffer et al. 2009).

However, restoration ecologists in the Southwestern United States have increasingly used the Plug and Pond method to restore riparian areas and optimal rangeland conditions. It can be used in any eroded channel to restore the channel to surface levels. This method includes excavating alluvial materials from flood plains, which forms ponds. The alluvial material is then used to plug incised channels. The plug stops sediment that is carried in the incised channel, upstream of the restoration site, and sediment back fills the channel restoring it to the floodplain or alluvial fan surface (Hammersmark 2008, 2009a, 2009b).

Hammersmark et al. (2008) have shown how such methods 1) increase the volume and storage capacity of groundwater; 2) decrease the magnitude of flood events; 3) increase the duration of flood plain inundation; 4) and decrease annual runoff and base flow. Subsequent studies by Hammersmark et al. (2009a, 2009b) have shown that the restoration of water tables also restores native plant species and community composition by allowing them to out-compete xeric, invasive and upland species that have invaded degraded sites. Monitoring of Plug and Pond restoration also indicated an increase in the spatial distribution of suitable habitat for mesic species (Hammersmark 2009a).

This type of abiotic restoration, coupled with reseeding of native species, can move systems toward more desirable states and even restore them to their former state. A study by Tate et al. (n.d.) has shown that Plug and Pond, as well as other natural channel design methods (Zeedyk & Clothier 2014), can not only restore nutrient-rich grasses, but increase
biomass productivity by over 200%. Mesic meadows adjacent to restored wetlands and riparian areas were shown to produce livestock with the highest weight gains and grasses with the consistently highest nutrient levels. Wet meadows produced livestock with weight gains 7% lower than mesic meadows and grass nutrients at a moderate level. Dry meadows produced livestock with weight gains 24% lower than mesic meadows and grass nutrient levels lower than both wet and mesic meadows (Tate et al. n.d.). This further illustrates the importance of water in delivering both biomass and nutrients to herbivores. The method has also been shown to be effective in arid environments in restoring channelized desert rangeland, desert seeps and washes.

While the Plug and Pond method was originally used for wetland restoration, practitioners in New Mexico, Texas and Arizona have also successfully used this technique for dryland restoration (Zeedyk & Clothier 2014). The building of plugs at specific locations in watersheds spreads the water back out across former “sheet flow” areas, deposits nutrients and sediment, and irrigates vegetation. This makes the landscape more productive for humans, livestock, plants and animals. It will also create a landscape that is more resilient to disturbance, which will become increasingly important as climate change persists. Perhaps most importantly, it also restores shallow groundwater levels and increases groundwater storage, which is critical to the survival of all the priority biodiversity features in the project area.

**Recommendations for Other NPI and CBMP Programs**

**Data collection.** The collection of good baseline data prior to project implementation is critical to calculating biodiversity gains and losses. This may seem simple, but it is often difficult to gather good baseline data. This is because projects that do not require lender
financing and projects developed by companies that do not have an internal commitment to NPI will rarely conduct biodiversity assessments before construction begins. Projects often change hands, as was the case with the OT mine site. This project was initiated by a much smaller mining company and later purchased by Rio Tinto. Although Rio Tinto has an internal commitment to NPI, the company also did not realize the benefit in conducting detailed baseline surveys. As a result, the entire mining operation was built without a full baseline assessment being done. OT later realized that it would need external funding to complete the extensive underground mineshaft they hoped to operate parallel to the open pit. This triggered international regulatory policies that required a baseline assessment be done, but it was too late.

The project still attempted to set baseline parameters for most species, but much of this data was incomplete and disturbances were underway. For this reason, among others, it is imperative that national governments develop their own NPI policies that require all new mines follow the same procedures, such as conducting a baseline assessment prior to construction.

New mining developments that require lender funding are also required to produce an ESIA that includes a rapid biodiversity assessment. These assessments are often done remotely and include extensive literature searches. Some ground-truthing is often done, but usually does not include extensive data collection or assessments of current conditions. Priority biodiversity features are set based on threatened and endangered species in the region and critical habitat for those species. However, I suggest that by focusing on T&E species and their habitat, the impacts the mine is having on abiotic features such as groundwater and erosion are easily overlooked. A mechanism should be put in place that 1)
requires abiotic mechanism, critical to the survival of all species, be included in core biodiversity monitoring, mitigation and offset strategies regardless of the location of the mine; and 2) explicitly compels monitoring teams to identify species, critical habitat, community dynamics and abiotic features that should be included in Biodiversity Action Plans and or Mitigation and Offset Strategies. For the OT project, these features would include monitoring impacts from erosion, mitigating erosion caused by infrastructure; monitoring waterholes and mitigating any impacts to them either direct or indirect; and monitoring whole raptor communities and/or whole rodent communities.

**Collaboration.** Developing a clear collaborative process is perhaps the most important part of a successful NPI program. There are many levels of collaboration that need to be developed and it is important that the collaborative process be set up before the program is implemented. This is because each step of the process will require the consensus of multiple stakeholders. The first tier of collaboration should be between the implementing organization (in our case, this was the Wildlife Conservation Society), the company (Oyu Tolgoi); the lenders (IFC & EBRD); the national government (Ministry of Environment and Green Development), and The Biodiversity Consultants (TBC). The implementing organization will be designing monitoring methodologies for the biodiversity features that TBC or a similar organization has developed. Monitoring methods will need to be approved by the lenders and TBC and funded by the mine project. The offset landscape will need to be approved by the national government.

Even if a country does not have an NPI policy, governments often have some sort of mitigation policy and have defined some sort of offset landscape where companies are required to do improvements or pay for existing conservation initiatives. However, NPI
programs designed to address the specific impacts of a given project sometimes focus on larger offset landscapes, are more specifically targeted toward impacted species or they address narrowly defined specific issues, such as the removal of fences. For this reason, it is very important to work with the national government to define offset landscapes that are relevant to the particular project. This should also include educating the government about offset design and implementation. Similarly, national governments and lenders should also be made aware of the cumulative impacts resulting from other mines in the impact area, as well as other human settlements and local practices that are also impacting biodiversity, so that methods for dealing with them can be jointly developed and approved.

The second tier of collaboration should be between the implementing organization, the mine, other mines, relevant conservation organizations, universities, other research and monitoring initiatives, parks and protected areas, and the local level government. Working with mining companies that are also in the impact area will assist project leaders in developing methodologies for monitoring cumulative impacts. Pooling resources can also increase the effectiveness of mitigation measures and similarly some mitigation measure will not be effective without the cooperation of all the parties responsible for impacts. Working with neighboring companies may also provide programs with additional data.

Most implementing organizations do not have the resources to conduct adequate biodiversity monitoring. It is therefore absolutely critical to develop good relationships with local universities, NGOs, protected areas and international and national research teams. These organizations and individuals can be contracted to help with the core monitoring and may also have preexisting data and experience that could be useful. Other conservation organizations and NGOs may be developing similar programs (which was the case with TNC
in Mongolia). Competition over who is doing what can slow the progress of important work, especially if other conservation organizations have close relationships with the national government in the project area. It is therefore best to make sure that goals are aligned and clear understandings developed before programs are implemented.

The third tier of collaboration should be between the implementing organization, the mine, adjacent communities and other community development initiatives. Mining projects, such as OT, often have community outreach departments that are working with communities on initiatives that range from new housing to rural livelihood development programs. These programs can easily conflict with biodiversity, mitigation and offset initiatives. For instance, while the biodiversity department and the CBMP were advocating for reduced livestock numbers in the Gobi, the OT community team was working with herders to increase their livestock herds.

Having a direct relationship with communities is also important in order to help communities understand what is being done on their behalf, what their impacts are on the environment and what the monitoring data has shown. For instance, communities may not blame OT for reductions in shallow groundwater if they understand how grazing and roads lead to erosion that lowers ground water tables. Arming communities with the information and tools they need to monitor their own grazing impacts helped the OT program to decrease negative sentiments toward the mine and helped herders to understand how to better manage their livestock. Similarly, equipping communities with methods for reducing erosion, especially around water holes, can not only reduce erosion and groundwater reductions, but also provides local people with another tool for reading the landscape and making wise decisions.
Outreach. For similar reasons, it is important to make sure that the larger public understands the mission and goals of NPI programs. Mining is always accompanied with negative sentiments from local and international communities. Making reports and findings available to the public can help to reduce mistrust and ease fears about the potential wholesale destruction of ecosystems near mine sites. It is important that everyone understands that it is not only mining that impacts landscapes, but all of the cumulative impacts that come with human settlements.

Data sharing and analysis. Lastly, it is important that a method for sharing data is developed before field seasons begin. This should include a repository where collaborators can access data independently. Internal agencies within the mining company should have scheduled data upload times so that researchers who need corresponding data for analysis are not required to schedule meetings at remote mine sites and wait for data to be organized and uploaded. Monitoring teams also need to be given ample time to analyze data. Reports are often due on a quarterly basis, leaving very little time for research teams to focus on analysis. We also discovered that equal amounts of time should be devoted to data analysis and data collection and that regular data analysis meetings should be scheduled to determine where working through data as a team is necessary. However, safeguards should be put in place to make sure researchers feel comfortable about sharing data. It is imperative that we share data with each other, with the government and other institutions and companies in a way that allows all participants to feel comfortable with the process.

Conclusion

The OT mitigation, offset and monitoring project is one of the largest and most comprehensive projects of its kind. It represents one of the best international environmental
policies in action in the world. The idea that development projects should demonstrate no net loss (NNL) of biodiversity from project impacts could significantly reduce land degradation if adopted by national governments. However, the policy still focuses on reducing threats to T&E species, similar to the U.S. Endangered Species Act (IFC 2012; EBRD 2014). If enforced solely by Equator Banks, such as the IFC and EBRD, the policy will only be applied to projects that require loans, similar to the U.S. Endangered Species Act, which only applies to projects using federal funds. Continued land degradation by Chinese-based or other extractive industries that do not need third party funding may undermine NNL/NPI projects, just as private development initiatives undermine endangered species protection in the U.S.

Similarly, focusing environmental policies on T&E species and their critical habitat does not address global or even regional declines in biodiversity. While we focus solely on the protection of threatened and endangered species, other species are rapidly becoming endangered due to continued anthropogenic forcings. A better model might include the protection of whole systems against land degradation that has cascading effects on regional biodiversity. As we have seen, abiotic processes are intricately linked to one another and require landscape-level protection. I suggest that NNL and NPI policies require that projects use global change threshold indicators as a means of prioritizing mitigation efforts.

For instance, Rio Tinto suggests that all of its projects will result in NNL of biodiversity. We have already exceeded global CO₂ concentrations, yet the OT project constructed a coal power plant to fuel its operations in Mongolia (Oyu Tolgoi 2012). Mongolia’s capital, Ulaanbaatar, has some of the worst air quality on the planet, due almost entirely to the use of coal (Amersaikhan et al. 2014). Therefore, the OT mine’s first priority
should have been to construct a solar power plant, which in the long run would have saved the project money as well as not added to growing atmospheric CO₂ concentrations. The situation is similar for land degradation. Guidelines for assessing the amount of land degraded due to anthropogenic forcings within the project impact area, and requiring that it be restored, may be more beneficial to wildlife than only focusing on T&E species.

The effectiveness and focus of restoration, mitigation, offset and monitoring programs often depend on the strengths of the project implementers. Each individual and each group of individuals has a focal set of expertise. When the full range of expertise needed on a project is not well-represented, key ecological processes and species can be overlooked. There is no point in blaming project implementers for this downfall because financial and human resource limitations will always lead to shortcomings. However, the creation of guidelines that map out how projects should identify priorities for biodiversity may help to ensure that critical human impacts are not overlooked. Similarly, the use of assessment tools that help to identify landscape-level impacts, similar to the Development by Design tool created by The Nature Conservancy, will strengthen mitigation, restoration and offset programs.
References


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Tables

Table 1: Elm Stand % Bare ground, Leaf Litter and Elm Cover 2014

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Figure 1: Map of existing and proposed mines in Mongolia (Mongolian Ministry of Mines 2013)
Figure 2: Map of the khulan range. The khulan range encompasses the entire project area. (WCS & SEA 2015)
Figure 3: Percentage of dead material on elm trees by site. Twelve trees in each site within four height classes were sampled. Each site represents the average of 12 trees.
Figure 4: Percentage of leaves eaten by insects. Twelve trees in each site within four height classes were sampled. Each site represents the average of 12 trees.
Figure 5: Average DBH of elm trees by site. Twelve trees in each site within four height classes were sampled. Each site represents the average of 12 trees.
Figure 6: Map of Mongolian railway lines and distribution of ungulates (WWF Mongolia 2011).
Chapter 5: Conclusion

Human induced land-use change creates a complex range of disturbances that cause worldwide biodiversity loss (WWF 2014). Large migratory herbivores are particularly susceptible to land-use change because of their relatively small population sizes compared to smaller bodied species (Calder 1984; Peters 1983) and their large home ranges (Lindstedt et al. 1986; Estes et al. 2011; Peters 1983). When access to critical resources, such as water and seasonal breeding and foraging ranges (Fynn & Bonyongo 2011), are blocked and resources degraded large die-off events can significantly reduce herbivore populations (Chase 2011). The fragmentation of critical habitat caused by the creation of new human developments such as roads, new communities, and industrial developments can lead to human-wildlife conflict and increased hunting, which further threatens wildlife populations (Elliot et al. 2008). This is because human-wildlife conflict often leads to the killing or relocating of problem animals and roads and humans living close to wildlife areas increases access for hunters (Radovani et al. 2015). Large industrial projects, such as mining, can have the biggest impact on species by introducing all of these disturbances.

In Chapter Two, I monitor the daily and hourly movement and assemblage patterns of large herbivores around water resources in Northern Botswana. This data is used to determine whether predictable patterns of resource use can help to mitigate human-wildlife conflict and reduce habitat fragmentation. Human-wildlife conflict is one the main threats to large herbivores in Africa (Elliot et al. 2008; Barnes 1996; Ogada et al. 2003) and conflicts are often a direct result of human caused habitat fragmentation and land-use change (Selier 2015; Ogada et al. 2003). The data showed that species access the river at specific times of day; that those times vary depending on season; that river morphology is correlated with
richness and abundance of herbivores; and that some species occupy specific niches in time and space to avoid competition for access to water.

The fact that species access water at specific intervals during each season provides an opportunity for humans accessing the same water points to avoid the riverfront and wetland during heavy use periods. It also informs farmers of the best times to monitor fields that are close to water (malapa farms) and may be opportunistically raided by herbivores. Perhaps most importantly, it provides village planners and conservation professionals with a tool for placing villages where they will have the least impact on large herbivores and designing movement corridors for herbivores that will be optimally used.

In Chapter Three, I explore how hunting bans can have the unintended consequence of exacerbating biodiversity loss and suggest a method for valuing wildlife resources in a way that helps to increase biodiversity and also supports rural communities. Using Botswana as a case study, I provide an example of how lack of local ownership in safari and trophy hunting industries has led to the establishment of secretive bushmeat markets. I explore the potential drivers of species loss and illustrate how the hunting ban has: led to loss of local livelihoods magnifying the need for illegal hunting; compelled people to obtain more livestock to increase their incomes, and displaced rural people leading to land-use change. I show how land-use change increases illegal hunting and human-wildlife conflict, fragments habitat, and blocks migratory routes, causing additional wildlife declines.

I then calculate the value of the wildlife resource in the Ngamiland District of Botswana. Calculations indicate that the wildlife resource in one district alone can be valued at over 1 billion USD (1,031,085,000 USD). I suggest that communities should be given ownership of the wildlife resource and that this will provide them with an incentive to
employ methods that increase wildlife populations the same as they increase their livestock herds. The value of wildlife can be used to leverage outside funding to create sovereign wealth funds, similar to Norway’s, that can assist communities in paying for the infrastructure needed to provide services such as: solar power cooperatives, grey and sewer water filtration systems and recycling, as well as livestock fencing and HWC mitigation.

Chapter 3, explores whether it is possible for large mining projects to result in no-net loss (NNL) of biodiversity or even have a net-positive impact (NPI) on biodiversity. We use the Oyu Tolgi mine, which is the world’s largest copper and gold mine as a case study. The mine is committed to having a net-positive impact on biodiversity by the time of mine closure and is mandated to do so by both internal and lender policies. As a result, the project has implemented a Core Biodiversity Monitoring Program (CBMP) to monitor its impacts. I examine the methods and results of this Program, provide lessons learned from the first two years of monitoring and outline best practices for similar programs that are in their development phase. I propose that focusing mitigation, offsets and monitoring solely on threatened and endangered species and their critical habitat may not be the most effective way to reduce biodiversity loss. I suggest that restoring and enhancing ecological processes and landscape-level degradation is the most effective mechanism for improving biodiversity; and that careful collaboration with program stakeholders is the key to a successful program.

Overall, it is clear that we can reduce our negative impacts on biodiversity through careful examination of the movement patterns and resource needs of the species we may be impacting. We can also reduce the degradation of whole landscapes by understanding how land is degraded and mitigating those impacts before they lead to the collapse of ecological process.
References


