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Research Article

Evaluating Conservation Effectiveness in a Tanzanian Community Wildlife Management Area

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ABSTRACT In Tanzania, community-based natural resource management (CBNRM) of wildlife occurs through wildlife management areas (WMAs). The WMAs consist of multiple villages designating land and managing it for wildlife conservation in return for a portion of subsequent tourism revenues. The ecological success or failure of WMAs for wildlife conservation is rarely quantified but is important for evaluating the efficacy of specific projects and the general concept of CBNRM. I used 3 analyses to evaluate the ecological effectiveness of wildlife conservation actions in the Burunge WMA. I compared wildlife and livestock densities inside and outside a WMA using 1 year of distance sampling data and compared wild and domestic ungulate densities before and after the implementation of management changes that increased wildlife protections within a subsection of Burunge WMA using 6 years of distance sampling surveys. I also compared giraffe (*Giraffa camelopardalis*) survival and population growth rate before and after the implementation of management changes that increased wildlife protections in a subsection of Burunge WMA using 5 years of photographic capture-recapture data. I found greater densities of wildlife and lower densities of livestock inside the WMA compared with outside. After the management changes, I documented significantly higher densities of several wild ungulate species and lower densities of domestic ungulates in the WMA. I found giraffe survival and population growth rate both increased in response to the management changes. Results indicated the WMA is effectively providing habitat and protection for wild ungulates while generally excluding domestic livestock. Ungulate densities, and giraffe survival and population growth rate over time indicated the management changes enacted in 2014–2015 resulted in positive effects for wild ungulates. These combined results indicate the ecological effectiveness of Burunge WMA and provide evidence that CBNRM can have positive effects on wildlife populations, particularly when support to grassroots law enforcement is provided. © 2018 The Wildlife Society.

KEY WORDS community-based natural resource management, conservation biology, ecological monitoring, impact assessment, ungulate, wildlife management area.

Community-based natural resource management (CBNRM) based on the transference of resource management and user rights to local communities has become one of the dominant paradigms of natural resource conservation, particularly in sub-Saharan Africa (Western and Wright 1994, Borgerhoff Mulder and Coppolillo 2005, Child and Barnes 2010, Nelson 2010). Unfortunately, the ecological effectiveness of CBNRM projects is only rarely assessed. A recent analysis reported only 13% of 159 CBNRM projects included quantification of ecological outcomes (Brooks et al. 2012). In Tanzania, CBNRM efforts for wildlife conservation decentralize use rights to local communities through the creation of wildlife management areas (WMAs). The WMAs are community-based conservation and development areas, with several villages setting aside land for wildlife conservation in return for most of the tourism revenues from these areas (Nelson 2010, United Republic of Tanzania 2012).

There have been several social and economic analyses of WMAs (Benjaminsen et al. 2013, Tetra Tech and Maliasili Initiatives 2013, Bluwstein et al. 2016, Moyo et al. 2016, Salerno et al. 2016), but data are scarce on the ecological effectiveness of WMAs for wildlife conservation (Lee and Bond 2018). Quantifying ecological effectiveness is important for evaluating specific projects and the general concept of CBNRM (Ferraro and Pressey 2015). These data are also important for determining whether site-specific management actions such as anti-poaching ranger patrols and habitat restoration and maintenance are achieving their intended goals. Burunge WMA in the Tarangire Ecosystem of northern Tanzania was one of the first WMAs operational in the country, formally created in March 2006. I am aware of no published data describing the ecological conservation

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effectiveness in Burunge WMA (Tetra Tech and Maliasili Initiatives 2013).

The objective of this observational study was to determine whether Burunge WMA was effective at conserving wildlife. I defined ecological effectiveness as positive responses of wildlife population densities, survival rate, or population growth rate. I predicted that wildlife densities would be greater inside Burunge WMA relative to outside, partially because the best extant wildlife habitat was included in Burunge WMA when it was formed, and partially because human activities had been restricted in Burunge WMA for approximately 6 years prior to my surveys.

Within the Little Chem Chem (LCC) subsection of Burunge WMA, management changes were implemented in 2014–2015 that increased wildlife, fuelwood, and forage resource protections. In 2014, Chem Chem Safaris began operating in the LCC hunting block subsection of Burunge WMA, replacing the previous sport hunting operations with photographic tourism and increased resource protection activities. In 2015, Protected Area Management Solutions (PAMS) Foundation began implementing their Tarangire-Manyara Protection Project to support community-based conservation throughout Burunge WMA. Both Chem Chem Safaris and PAMS Foundation trained and equipped rangers to conduct anti-poaching activities protecting wildlife and fuelwood, and to prevent livestock encroachment. I predicted that when I compared ungulate densities in the LCC subsection between periods after (i.e., 2015–2017) and before (i.e., 2012–2014) management changes, I would observe resident wildlife densities increase, and livestock densities decline in the after period, indicating ecological success of the management actions initiated in 2014–2015. Finally, I predicted giraffe (*Giraffa camelopardalis*) survival and population growth rate should increase after the management changes in LCC, relative to the before period.

STUDY AREA

Burunge WMA is 226 km^2 (excluding Lake Burunge) and located between latitude 3.94°S to 3.66°S and longitude 35.73°E to 35.97°E in the Tarangire ecosystem, with Tarangire National Park to the southeast, Lake Manyara National Park to the northwest, and village lands to the northeast and southwest (Fig. 1). Elevation is 1,000 m with generally flat topography, and mean total annual rainfall of 650 mm is distributed in 3 precipitation seasons (short rains, long rains, and dry; Foley and Faust 2010). Vegetation inside the WMA is a mix of acacia (*Acacia* spp.) woodland, riverine vegetation, and edaphic grassland (Lamprey 1964), with areas outside the WMA predominately smallholding farms and edaphic grassland. Dominant native wildlife fauna are a diverse assemblage of ungulates and predators (Lamprey 1964). The LCC hunting block area is a 32-km^2 subsection of Burunge WMA stretching northeast from the shore of Lake Burunge bordered on the north and west by the Tarangire River, and on the east by Tarangire National Park (Fig. 1). I used data I collected from 2012 to 2017. Details of WMA history and management structure are given elsewhere (Benjaminsen et al. 2013, Tetra Tech and Maliasili Initiatives 2013, Moyo et al. 2016, Salerno et al. 2016).

METHODS

I used 3 analyses to evaluate the ecological effectiveness of wildlife conservation actions in the Burunge WMA. First, I used 1 year (2016) of distance sampling transect surveys for larger-sized wildlife (<1 kg) and livestock in and around Burunge WMA to quantify wildlife and livestock densities inside versus outside Burunge WMA as an index of differences in wildlife habitat quality and land use. Second, I used 6 years (2012–2017) of distance sampling transect surveys for wild and domestic ungulates within the LCC subsection of Burunge WMA to describe changes in densities as responses to management changes. I chose to monitor ungulate species over the long term because they are important to the photographic ecotourism industry in Tanzania. Third, I used 5 years of giraffe photographic capture-recapture data (2012–2016) from the LCC hunting block to evaluate whether survival and population growth rate of a large-bodied, resident species that was never legally hunted in the area were affected by the management changes initiated in 2014–2015.

This research was conducted with permission from the Tanzania Commission for Science and Technology, the Tanzania Wildlife Research Institute, Tanzania National Parks, Burunge Wildlife Management Area, and the villages therein. No Institutional Animal Care and Use Committee permit was required because the study was entirely observational and included no manipulations of animals or their habitats.

Transect Surveys

I conducted animal surveys using distance sampling methods (Buckland et al. 2001, 2015) during 3 sampling occasions/year near the end of each precipitation season. In each sampling occasion I surveyed fixed-route transects along dirt vehicle pathways in the study area using a Toyota Land Cruiser (Toyota, Aichi, Japan) with a pop top. Driving speed was maintained between 15 kph and 20 kph on all transect routes, and all surveys included the same 2 observers (1 standing and 1 seated) to minimize among-observer variation in detection probabilities, and a driver. Observers sampled survey routes 2 times in every sampling occasion with replicates separated by 1 week.

In 2016, I surveyed all transects inside (64 km) and outside (44 km) of Burunge WMA (Fig. 1) 6 times (3 sampling occasions $\times$ 2 replicates/occasion) to compare larger-sized wildlife and domestic ungulate densities in and out of the WMA. From January 2012 to October 2017, I surveyed transects 36 times (6 yr $\times$ 3 sampling occasions/yr $\times$ 2 replicates/occasion) in the LCC hunting block area (24 km) to compare wild and domestic ungulate densities and giraffe survival and population growth rate before and after the major management changes in that subsection of Burunge WMA. My study design implemented identical replicate surveys to quantify relative densities. Buckland et al. (2001, 2015) recommend systematic random designs, ≥10

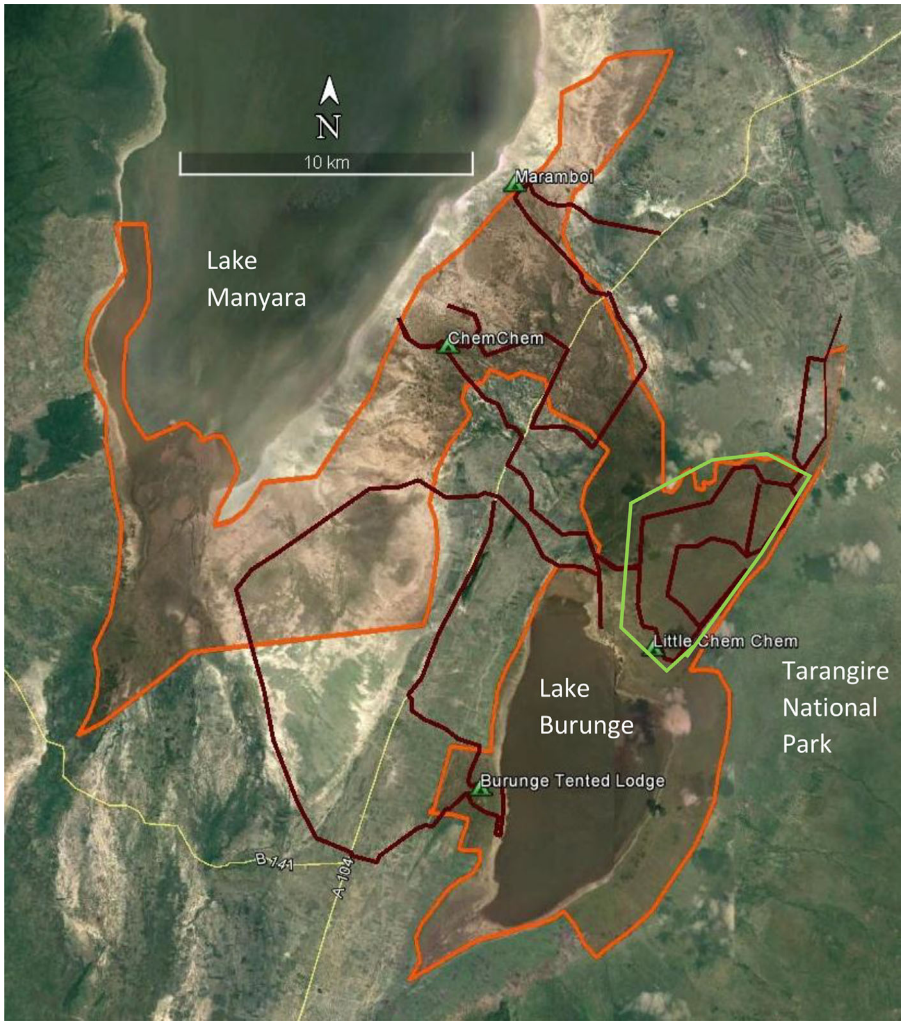

Figure 1. Burunge Wildlife Management Area (WMA; orange outline and fill; data courtesy Burunge WMA), Little Chem Chem hunting block subsection (green outline), main tourism lodges (tent symbols), main roads (yellow lines), and wildlife and livestock survey routes (dark red lines), Tanzania, 2012–2017. Tarangire National Park is to the southeast, Lake Manyara National Park is to the northwest, village lands are to the northeast and southwest. The satellite imagery is from Google Earth (Google, Mountain View, CA, USA).

replicate transect lines, and $\geq$60 observations for studies whose objective is to estimate true density of an area. My design did not entirely conform to these recommendations (6 and 36 transects), but my design of systematic replicated surveys was appropriate for my purposes of analyzing relative densities inside versus outside the WMA, and before versus after management changes.

I collected distance data for all larger-sized (>1 kg) mammal wildlife species and common ostrich (*Struthio camelus*) visible along both sides of the vehicle pathway out to 500 m. Distance data records the group size and perpendicular distance from the transect to each group of animals when first detected. When an observer sighted a group or singleton, I halted the vehicle and recorded the perpendicular distance from the vehicle pathway to the animal(s) measured with a laser rangefinder, the number of individuals, and the global positioning system (GPS) position of the vehicle. If the sighting was a cluster of animals, I measured the perpendicular distance from the vehicle pathway to the middle of the group.

Whenever I encountered giraffes during distance sampling, I collected giraffe photographic capture-recapture data. I either marked or recaptured all individuals by slowly approaching and photographing the animal's right side. I photographed and later identified individual giraffes using their unique and unchanging coat patterns (Foster 1966). I also recorded sex, GPS location, and age class. I categorized giraffes into 4 age classes: newborn calf (0–3 months old), older calf (4–11 months old), subadult (1–3 yrs old for females, 1–6 yr old for males), or adult (>3 yr for females, >6 yr for males) using a suite of physical characteristics, including body shape, relative length of the neck and legs, ossicone characteristics (Strauss et al. 2015), and height (Lee et al. 2016). I matched giraffe identification images using WildID (Bolger et al. 2012), a computer program that matched a large test dataset of giraffe images collected using my protocols with a very low false rejection rate (0.007) and 0.0 false acceptance rate (Bolger et al. 2012). Based on matching results, I created individual encounter histories for all giraffes for analysis.

I documented 16 larger-sized wildlife species, including olive baboons (*Papio anubis*), banded mongoose (*Mungos mungo*), cape buffalo (*Syncerus caffer*), bushbuck (*Tragelaphus scriptus*), Kirk's dik-dik (*Madoqua kirkii*), African elephant (*Loxodonta africana*), giraffe, impala (*Aepyceros melampus*), lesser kudu (*Tragelaphus imberbis*), bohor reedbuck (*Redunca redunca*), Thomson's gazelle (*Eudorcas rufifrons*), warthog (*Phacochoerus africanus*), common waterbuck (*Kobus ellipsiprymnus*), wildebeest (*Connochaetes taurinus*), and plains zebra (*Equus quagga*). I observed 4 species of domestic ungulate: cattle, donkeys, sheep, and goats. I combined sheep and goats because they are usually herded together in large groups, and differentiation is unreliable when the groups are not very near to the observer.

Distance Data Analysis

I analyzed distance data for each species with >9 observations separately with program DISTANCE 6.0 (Thomas et al. 2010) to estimate density of animals in each site while accounting for variation in detectability according to distance from the road transect. For rare species (<10 observations), I computed a seasonal encounter rate/km of transect surveyed and computed density as for belt transects 1,000 m wide, centered on the path I traveled. I analyzed distance data following recommendations in Buckland et al. (2015). I considered all vehicle pathways surveyed within a site during a single sampling event as a single transect, and treated each of the survey occasions as replicate samples. I discarded the farthest 15% of observations (Buckland et al. 2015). I plotted frequency histograms of perpendicular distances and fitted models to the histogram based on the key function and series expansion approach (Buckland et al. 2015). I fitted uniform, half-normal, and hazard-rate key functions with cosine and simple polynomial series expansions (Buckland et al. 2015). I fitted the key function models and associated series expansions to the data and used corrected Akaike's Information Criterion (AIC_c) to select the best detection function model (Buckland et al. 2015). I assessed goodness of fit of the top model using chi-square and Cramer von Misses tests (Buckland et al. 2015). I regressed the logarithm of cluster size against the detection probability and adjusted detectability based on the expected cluster size (Buckland et al. 2015).

I estimated site-specific densities inside and outside Burunge WMA, and year-specific densities within the LCC subsection of Burunge WMA, using the top-ranked species-specific models, or belt transect estimates for rare species. I used z-tests to compare wildlife and livestock densities inside and outside Burunge WMA from 2016 data (Buckland et al. 2001, 2015). To describe the changes in densities in the LCC block after the management changes in 2014–2015, I split the data into before (2012–2014) and after (2015–2017) periods and compared the 2 periods using z-tests (Buckland et al. 2001, 2015).

Giraffe Survival and Population Growth Rate

I modeled and estimated seasonal (3 seasons/yr) giraffe survival and population growth rate in Program MARK 7.1 (White and Burnham 1999). I modeled temporal variability in apparent survival (ϕ) and realized population growth rate (λ) using the robust design Pradel (Huggins closed capture) model (Pollock 1982, Pollock and Otto 1983, Pradel 1996) with data from all age classes. The Pradel model estimates ϕ and λ along with nuisance parameters of capture (p) and recapture (c).

I used Program MARK to evaluate goodness of fit by estimating overdispersion using a median $\hat{c}$ procedure (White and Burnham 1999). Values of $\hat{c} > 1$ indicate some overdispersion in the data, and if the $\hat{c} > 3$, the variance is inflated by $\hat{c}$. Throughout model ranking and selection procedures, I ranked models using AIC_c and used model AIC_c weights (w_i) as a metric for strength of evidence supporting a given model as the best description of the data (Burnham and Anderson 2002). The primary parameters of interest were ϕ and λ, so I used a 2-phase *a priori* approach to model development by first evaluating models of capture (p) and recapture (c) probabilities, and then modeling the parameters of interest (Lebreton et al. 1992, Anthony et al. 2006).

To test the predictions that apparent survival and population growth rate were higher after the management changes in 2014–2015, I compared a model where ϕ and λ differed between the 2 time periods of before (2012–2014) and after (2015–2016), with a constant model where ϕ and λ did not change over time, and intermediate models (see below). The before-after-control-impact (BACI) study design provided strong statistical inference (Green 1979, Stewart-Oaten et al. 1986, Underwood 1992). I defined the impact in this case as the changes in management. To account for possible temporal changes in ϕ and λ due to external environmental variation, I included data from a control site in adjacent Tarangire National Park where I collected giraffe encounter history data according to the same protocols as I used in Burunge WMA (Lee et al. 2016). I expected no difference in ϕ and λ in Tarangire between the before and after time periods, but I was able to explicitly examine this assumption during model selection.

Once I obtained the most parsimonious model structure for p and c, I ranked models of ϕ and λ to determine whether those parameters changed between the before and after periods using a BACI design. I did this by ranking nested models of ϕ and λ as site- and period-dependent {ϕ(site $\times$ period), λ(site $\times$ period)}, as constants (constant), as site-specific (site), and as an additive combination of site and period (site + period). I used model averaging (Burnham and Anderson 2002) to estimate values of ϕ and λ. Model averaging has the advantage of incorporating uncertainty when multiple models are competing (Burnham and Anderson 2002) and produces a more stable set of parameter estimates (Doherty et al. 2012).

RESULTS

Distance Data

In 2016, I collected 305 observations of wildlife and 344 observations of livestock during distance sampling inside and outside Burunge WMA (Table 1). From 2012–2017, I

Table 1. Number of observations, mean site-specific densities (number/km^2), standard errors (SE), and statistical results for differences in livestock and wildlife densities between Burunge Wildlife Management Area (BWMA) and outside the BWMA, Tanzania, 2016.

Species	Number of observations	BWMA (number/km^2)	SE	Outside (number/km^2)	SE	Diff[a]	z-score[b]	P
Cattle	165	9.39	0.59	31.88	3.90	−22.49	5.71	<0.001
Donkeys	33	0.59	0.32	1.09	0.36	−0.50	1.04	0.149
Sheep and goats	146	8.58	2.97	35.21	1.60	−26.63	7.89	<0.001
Olive baboon	3	0.12	0.12	0.00	0.00	0.12	1.00	0.159
Banded mongoose	1	0.03	0.03	0.00	0.00	0.03	1.00	0.159
Buffalo	13	0.18	0.09	0.00	0.00	0.18	2.04	0.021
Bushbuck	1	0.01	0.01	0.00	0.00	0.01	1.00	0.159
Dik-dik	57	0.41	0.11	0.05	0.01	0.36	3.33	<0.001
Elephant	14	0.48	0.11	0.00	0.00	0.48	4.33	<0.001
Giraffe	21	0.48	0.16	0.04	0.04	0.45	2.77	0.003
Impala	61	1.71	0.34	0.03	0.03	1.68	4.91	<0.001
Lesser kudu	2	0.01	0.01	0.00	0.00	0.01	2.00	0.023
Ostrich	4	0.01	0.01	0.09	0.05	−0.08	1.46	0.070
Reedbuck	3	0.05	0.02	0.00	0.00	0.05	1.96	0.025
Thomson's gazelle	12	0.56	0.24	0.09	0.05	0.47	1.90	0.029
Warthog	4	0.05	0.05	0.03	0.03	0.03	0.46	0.323
Waterbuck	6	0.10	0.06	0.00	0.00	0.10	1.67	0.047
Wildebeest	36	3.45	0.63	0.29	0.27	3.15	4.58	<0.001
Zebra	67	4.66	1.80	0.40	0.13	4.26	2.36	0.009

[a] BWMA–Outside.
[b] Diff/SE for the diff, where SE for the diff $= \sqrt{\mathrm{SE}^2_{\mathrm{BWMA}} + \mathrm{SE}^2_{\mathrm{Outside}}}$.

collected 616 observations of wild ungulates and 31 observations of livestock during distance sampling in the LCC hunting block subsection of Burunge WMA (Table 2). All detection functions passed goodness-of-fit tests, so I relied upon AIC$_c$ to select the best function and used the top-ranked detection functions to estimate densities for each species.

In comparing the wildlife and livestock densities inside and outside Burunge WMA in 2016, 11 of the 16 species of wildlife observed had significantly greater densities inside the WMA relative to outside (Table 1). Sheep and goats combined and cattle had significantly lower densities inside the WMA relative to outside (Table 1). Ostrich was the only species of wildlife with greater density outside the WMA, but the difference was not significant (Table 1).

To examine ungulate densities before and after management changes in the LCC block, I needed sufficient numbers of observations in every year to estimate annual densities. Four species of wild ungulates, cattle, and sheep and goats combined had sufficient data to estimate annual densities and examine differences before and after management changes were implemented in 2014–2015 (Table 2). Two of 4 wildlife species' densities increased significantly, and all livestock species densities decreased significantly (Fig. 2, Table 2), indicating ecological success of the management changes that occurred in 2014–2015.

Giraffe Survival and Population Growth Rate

The photographic capture-recapture data for giraffe resulted in 359 individual encounter histories in the LCC hunting block subsection of Burunge WMA, and 1,090 encounter histories from Tarangire National Park. I detected minor overdispersion in the data ($\hat{c} = 1.19$, 95% CI = 1.17–1.21), but because the computed $\hat{c}$ adjustment was <3, I did not apply a variance inflation factor (Burnham and Anderson 2002, Choquet et al. 2005).

Capture and recapture parameters were best modeled by full site- and time-dependence, so for all subsequent model-selection procedures, the parameters p and c remained always in this structure: {p(site × period), c(site × period)}. Model selection for ϕ and λ indicated there was evidence for period-specific variation in survival and population growth rate in Burunge WMA (Table 3). Model-averaged parameter estimates indicated increases in apparent survival and

Table 2. Mean period-specific densities (number/km^2), standard errors (SE), and statistical results for differences in livestock and wildlife densities before (2012–2014) and after (2015–2017) major management changes in the Little Chem Chem hunting block subsection of Burunge Wildlife Management Area, Tanzania.

Species	Number of observations	Before (number/km^2)	After (number/km^2)	SE Before	SE After	Diff[a]	P[b]
Buffalo	23	0.026	0.241	0.01	0.05	0.215	0.022
Dik-dik	208	0.379	0.612	0.07	0.16	0.233	0.071
Impala	277	1.614	4.294	0.50	0.71	2.680	0.035
Waterbuck	55	0.171	0.395	0.05	0.08	0.224	0.092
Cattle	21	2.195	0.173	0.37	0.05	−2.022	0.014
Sheep and goats	10	0.973	0.118	0.25	0.05	−0.854	0.029

[a] After–Before.
[b] P based on z-score, where z-score = Diff/SE for the diff, where SE for the diff $= \sqrt{\mathrm{SE}^2_{\mathrm{Before}} + \mathrm{SE}^2_{\mathrm{After}}}$.

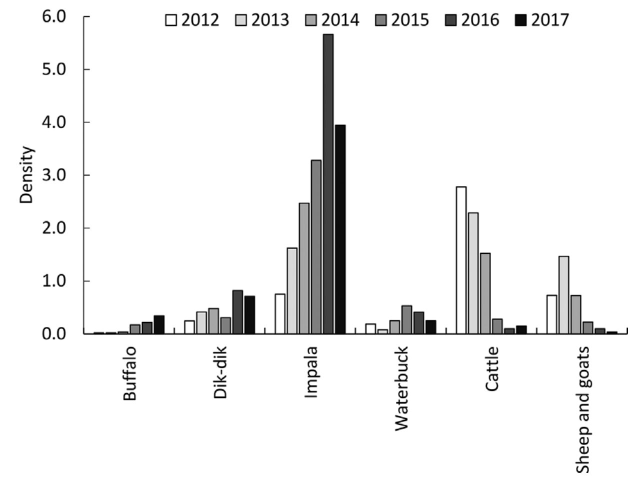

Figure 2. Densities (number/km^2) of wildlife increased over time within the Little Chem Chem hunting block area of Burunge Wildlife Management Area, Tanzania, 2012–2017. This area was a commercial hunting operation in 2012 and transitioned to photographic wildlife tourism with wildlife protection support from Protected Area Management Solutions (PAMS) Foundation and Chem Chem Safaris in 2014–2015.

population growth rate after management changes to increase wildlife protections were put into effect (Fig. 3). These increases were not present in the control site in Tarangire National Park, indicating they were the result of the impact of management changes in Burunge WMA.

DISCUSSION

I found clear evidence of ecological effectiveness (as I defined it) in Burunge WMA from all 3 of my analyses. First, Burunge WMA contained significantly higher densities of wildlife relative to adjacent village lands outside the WMA, and lower densities of livestock. Second, densities of wild ungulates increased and livestock densities decreased within the LCC hunting block area after changes in management that increased resource protections were enacted there.

Table 3. Model selection results for before-after-control-impact assessment of apparent survival (ϕ) and population growth rate (λ) changes due to management changes that increased wildlife protections (i.e., impact) in the Little Chem Chem hunting block subsection of Burunge Wildlife Management Area relative to the control site in Tarangire National Park 2012–2016, Tanzania. I ranked models by difference in corrected Akaike's Information Criterion (ΔAIC_c) and provide AIC_c weight (w_i) and number of parameters (K) for each model.

Model[a]	ΔAIC_c	w_i	K	Deviance
ϕ (site × period), λ (site × period)	0.00	0.36	102	37,000.07
ϕ (site + period), λ (site × period)	0.09	0.34	100	37,004.27
ϕ (site), λ (site × period)	1.66	0.16	98	37,009.94
ϕ (site × period), λ (site + period)	1.89	0.14	100	37,006.06
ϕ (site × period), λ (site)	17.27	0.00	98	37,025.55
ϕ (site × period), λ (constant)	25.55	0.00	97	37,035.88
ϕ (constant), λ (site × period)	38.54	0.00	97	37,048.87

[a] Period indicates parameter estimates varied between before (2012–2014) and after (2015–2016), site indicates the control and impact sites had different estimates, and constant indicates constant estimate (no time or site dependence).

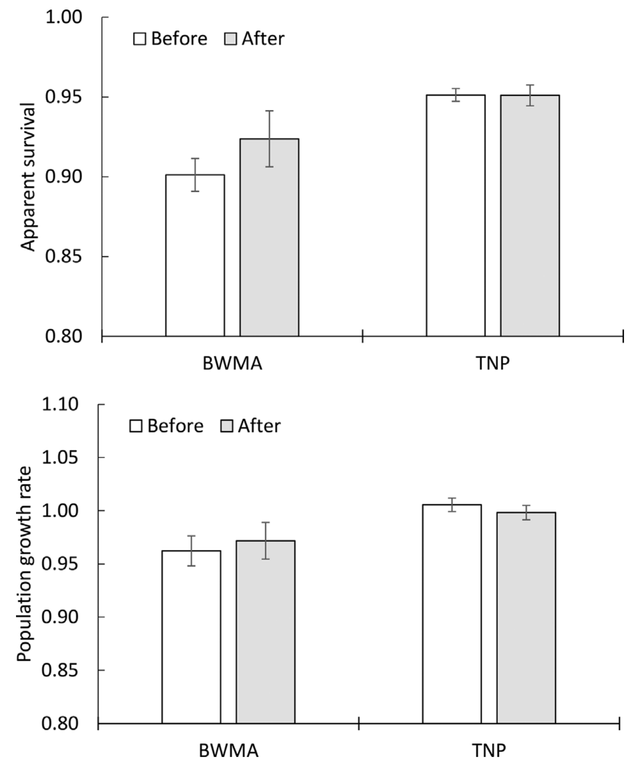

Figure 3. Model-averaged parameter estimates of seasonal (3 seasons/yr) giraffe apparent survival and population growth rate before and after management changes that increased wildlife protections in Burunge Wildlife Management Area (BWMA), relative to a control site in Tarangire National Park (TNP), indicate both parameters increased as a result. I collected data in 2012–2016 in the Tarangire ecosystem, Tanzania. Error bars indicate standard errors.

Third, apparent survival and population growth rate of giraffe in the LCC block area increased after the changes in management there, relative to a control site in Tarangire National Park. Each of these results indicate success of the Burunge WMA plan for wildlife conservation.

Very few wildlife were detected outside Burunge WMA, with the exception of ostrich. The differences in wildlife densities inside Burunge WMA versus outside is, in part, due to differences in the dominant vegetation communities. Most of the land outside Burunge WMA is farmland or open grassland with little tree or shrub cover, whereas the densest remaining natural woody vegetation in the study area is found inside Burunge WMA. This suggests that the land-use planning for wildlife when the WMA was created was appropriately designated to maximize conservation of remaining woody vegetation, and that protection efforts to preserve the woody plants have been effective at minimizing wood cutting.

The success record of CBNRM in general is highly variable (Kellert et al. 2000, Tallis et al. 2008, Brooks et al. 2012, Brooks 2017), and the socio-economic performance of WMAs has been criticized (Benjaminsen et al. 2013, Bamford et al. 2014, Bluwstein et al. 2016, Moyo et al. 2016). However, evidence is beginning to indicate that positive social and ecological outcomes can result from WMA projects (Tetra Tech and Maliasili Initiatives 2013, Pailler et al. 2015, Salerno et al. 2016, Lee and Bond 2018). My data have demonstrated that WMA establishment and management in Tanzania as practiced in Burunge WMA had positive outcomes for wildlife species densities and demographic rates.

The positive ecological effects I documented in Burunge WMA add to the evidence that CBNRM can be a successful strategy for natural resource conservation (Child and Barnes 2010, Brooks et al. 2012, Brooks 2017). The ecological success of Burunge WMA is likely linked to its age, the large ecotourism industry in Tanzania, and Burunge WMA's location in the Tarangire ecosystem that includes proximity to 2 popular national parks on the main tourism circuit (Brooks et al. 2012, Brooks 2017). Additionally, the village game scouts and management of Burunge WMA were supported by training, technical assistance, and capacity building, all important factors in CBNRM project success (Brooks 2017).

Despite the apparent positive conservation outcome, my results should not be construed to indicate that current efforts are sufficient to sustain Burunge WMA in the longer term. Populations of livestock and resident wildlife must be continuously assessed, so management actions can be evaluated (Borgerhoff Mulder et al. 2007). The high variability of wildlife population densities over time requires regular monitoring (Link et al. 1994), and monitoring should always be related to nearby areas of similar vegetation community types (Lee and Bond 2018). Locally based monitoring schemes should be encouraged (Schuette et al. 2018), as this can reinforce community-led resource management systems and lead to more sustainable community-based conservation (Danielsen et al. 2005).

MANAGEMENT IMPLICATIONS

The training and support of village rangers to conduct anti-poaching activities and prevent livestock encroachment is resulting in greater wildlife densities and lower livestock densities, so I suggest these activities continue to build and maintain capacity of Burunge WMA staff. Regular monitoring should be conducted and locally based monitoring schemes should be encouraged.

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LITERATURE CITED

Anthony, R. G., E. D. Forsman, A. B. Franklin, D. R. Anderson, K. P. Burnham, G. C. White, C. J. Schwarz, J. D. Nichols, J. E. Hines, G. S. Olson, and S. H. Ackers. 2006. Status and trends in demography of northern spotted owls, 1985–2003. Wildlife Monographs 163:1–48.

Bamford, A. J., D. Ferrol-Schulte, and J. Wathan. 2014. Human and wildlife usage of a protected area buffer zone in an area of high immigration. Oryx 48:504–513.

Benjaminsen, T. A., M. J. Goldman, M. Y. Minwary, and F. P. Maganga. 2013. Wildlife management in Tanzania: state control, rent seeking and community resistance. Development and Change 44:1087–1109.

Bluwstein, J., F. Moyo, and R. Kicheleri. 2016. Austere conservation: understanding conflicts over resource governance in Tanzanian wildlife management areas. Conservation and Society 14:218–231.

Borgerhoff Mulder, M., T. Caro, and O. A. Msago. 2007. The role of research in evaluating conservation strategies in Tanzania: the case of the Katavi-Rukwa ecosystem. Conservation Biology 21:647–658.

Borgerhoff Mulder, M., and P. Coppolillo. 2005. Conservation: linking ecology, economics, and culture. Princeton University Press, Princeton, New Jersey, USA.

Brooks, J. S. 2017. Design features and project age contribute to joint success in social, ecological, and economic outcomes of community-based conservation projects. Conservation Letters 10:23–32.

Brooks, J. S., K. A. Waylen, and M. Borgerhoff Mulder. 2012. How national context, project design, and local community characteristics influence success in community-based conservation projects. Proceedings of the National Academy of Sciences 109:21265–21270.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling estimating abundance of biological populations. Oxford University Press, Oxford, United Kingdom.

Buckland, S. T., E. A. Rexstad, T. A. Marques, and C. S. Oedekoven. 2015. Distance sampling: methods and applications. Springer, New York, New York, USA.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and inference: a practical information theoretic approach. Springer, New York, New York, USA.

Child, B., and G. Barnes. 2010. The conceptual evolution and practice of community-based natural resource management in southern Africa: past, present and future. Environmental Conservation 37:283–295.

Choquet, R., A. M. Reboulet, J. D. Lebreton, O. Gimenez, and R. Pradel. 2005. U-CARE 2.2 user's manual. CEFE, UMR 5175. CNRS, Montpellier, France.

Danielsen, F., N. D. Burgess, and A. Balmford. 2005. Monitoring matters: examining the potential of locally-based approaches. Biodiversity Conservation 14:2507–2542.

Doherty, P. F., G. C. White, and K. P. Burnham. 2012. Comparison of model building and selection strategies. Journal of Ornithology 152-(Supplement 2):S317–S323.

Ferraro, P. J., and R. L. Pressey. 2015. Measuring the difference made by conservation initiatives: protected areas and their environmental and social impacts. Philosophical Transactions of the Royal Society B: Biological Sciences 370:20140270.

Foley, C. A. H., and L. J. Faust. 2010. Rapid population growth in an elephant *Loxodonta africana* population recovering from poaching in Tarangire National Park, Tanzania. Oryx 44:205–212.

Foster, J. B. 1966. The giraffe of Nairobi National Park: home range, sex ratios, the herd, and food. East African Wildlife Journal 4:139–148.

Green, R. H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley and Sons, Hoboken, New Jersey, USA.

Kellert, S. R., J. N. Mehta, S. A. Ebbin, and L. L. Lichtenfeld. 2000. Community natural resource management: promise, rhetoric, and reality. Society and Natural Resources 13:705–715.

Lamprey, H. 1964. Estimation of the large mammal densities, biomass, and energy exchange in the Tarangire game reserve and the Maasai steppe in Tanzania. East African Wildlife Journal 1:3–92.

Lebreton, J.-D., K. P. Burnham, J. Clobert, and D. R. Anderson. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67–118.

Lee, D. E., and M. L. Bond. 2018. Quantifying the ecological success of a community-based wildlife conservation area in Tanzania. Journal of Mammalogy 99:459–464.

Lee, D. E., M. L. Bond, B. M. Kissui, Y. A. Kiwango, and D. T. Bolger. 2016. Spatial variation in giraffe demography: a test of two paradigms. Journal of Mammalogy 97:1015–1025.

Link, W. A., R. J. Barker, J. R. Sauer, and S. Droege. 1994. Within-site variability in surveys of wildlife populations. Ecology 75:1097–1108.

Moyo, F., J. Ijumba, and J. F. Lund. 2016. Failure by design? Revisiting Tanzania's flagship Wildlife Management Area Burunge. Conservation and Society 14:232–242.

Nelson, F. 2010. Community rights, conservation and contested land: the politics of natural resource governance in Africa. Earthscan, London, United Kingdom.
Pailler, S., R. Naidoo, N. D. Burgess, O. E. Freeman, and B. Fisher. 2015. Impacts of community-based natural resource management on wealth, food security and child health in Tanzania. PLoS ONE 10:e0133252. https://doi.org/10.1371/journal.pone.0133252
Pollock, K. H. 1982. A capture-recapture design robust to unequal probability of capture. Journal of Wildlife Management 46:752–757.
Pollock, K. H., and M. C. Otto. 1983. Robust estimation of population size in closed animal populations from capture-recapture experiments. Biometrics 39:1035–1049.
Salerno, J., M. B. Mulder, M. N. Grote, M. Ghiselli, and C. Packer. 2016. Household livelihoods and conflict with wildlife in community-based conservation areas across northern Tanzania. Oryx 50:702–712.
Schuette, P., N. Namukonde, M. S. Becker, F. G. Watson, S. Creel, C. Chifunte, W. Matandiko, P. Millhouser, E. Rosenblatt, and C. Sanguinetti. 2018. Boots on the ground: in defense of low-tech, inexpensive, and robust survey methods for Africa's under-funded protected areas. Biodiversity Conservation 27:2173–2191.
Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. Environmental impact assessment: pseudoreplication in time? Ecology 67:929–940.
Strauss, M. K. L., M. Kilewo, D. Rentsch, and C. Packer. 2015. Food supply and poaching limit giraffe abundance in the Serengeti. Population Ecology 57:505–516.
Tallis, H., P. Kareiva, M. Marvier, and A. Chang. 2008. An ecosystem services framework to support both practical conservation and economic development. Proceedings of the National Academy of Sciences 105:9457–9464.
Tetra Tech and Maliasili Initiatives. 2013. Tanzania Wildlife Management Areas evaluation. USAID, Dar es Salaam, Tanzania.
Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. Bishop, T. A. Marques, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5–14.
Underwood, A. J. 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology 161:145–178.
United Republic of Tanzania. 2012. The Wildlife Conservation (Wildlife Management Areas) Regulations. Ministry of Natural Resources and Tourism. Government Printer, Dar es Salaam, Tanzania.
Western, D. and R. M. Wright. 1994. Natural connections: perspectives in community-based conservation. Island Press, Washington, D.C., USA.
White, G. C., and K. P. Burnham. 1999. Program MARK: survival rate estimation from both live and dead encounters. Bird Study 46: S120–S139.

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