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Quantifying the ecological success of a community-based wildlife conservation area in Tanzania

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In Tanzania, community-based natural resource management of wildlife occurs through the creation of Wildlife Management Areas (WMAs). WMAs consist of multiple villages designating land for wildlife conservation, and sharing a portion of subsequent tourism revenues. Nineteen WMAs are currently operating, encompassing 7% of Tanzania's land area, with 19 more WMAs planned. The ecological success or failure of WMAs for wildlife conservation has yet to be quantified. We defined ecological success in this case as significantly greater densities of wildlife and significantly lower densities of livestock in the WMA relative to the control site, after the WMA was established. We used 4 years of distance sampling surveys conducted 6 times per year for wild and domestic ungulates to quantify wildlife and livestock densities before and after the establishment and implementation of management efforts at Randilen WMA, relative to a control site on adjacent land of similar vegetation and habitat types. We documented similarity between the sites before WMA establishment, when both sites were managed by the same authority. After WMA establishment, we documented significantly higher densities of resident wildlife (giraffes and dik-diks) and lower densities of cattle in the WMA, relative to the control site, indicating short-term ecological success. Continued monitoring is necessary to determine longer-term effects, and to evaluate management decisions.

Key words: before-after-control-impact, community-based natural resource management, environmental assessment, ungulate, wildlife management area, wildlife monitoring

Community-based natural resource management (CBNRM), established on the transference of resource management and user rights to local communities, is promoted as a conservation tool, and has become the dominant paradigm of natural resource conservation (Western and Wright 1994; Borgerhoff Mulder and Coppolillo 2005; Nelson 2010). Unfortunately, the ecological success or failure of CBNRM projects is rarely rigorously assessed. In a recent meta-analysis, only 13% of 159 CBNRM projects included quantification of ecological outcomes (Brooks et al. 2012).

In Tanzania, CBNRM efforts to decentralize wildlife management to local communities occur through the creation of Wildlife Management Areas (WMAs). WMAs are community-based conservation and development areas, with several villages setting aside land for wildlife conservation in return for the majority of tourism revenues from these areas (Nelson 2010; URT 2012). Nineteen WMAs are currently operating, encompassing 7% (6.2 million ha) of Tanzania's land area, with 19 more WMAs planned. Each WMA is independently managed. Most CBNRM programs are aimed at promoting conservation while maintaining or improving people's standards of living, but most CBNRM programs have had only limited success at achieving both conservation and human development goals (Newmark and Hough 2000). There have already been social and economic critiques of WMAs, such as onerous bureaucratic demands, continued governmental control over revenue collection, the large role played by outside conservation organizations, and economic and human rights failures (Goldman 2003; Igoe and Croucher 2007; Benjaminsen et al. 2013). However, the ecological value or success of WMAs for wildlife conservation has yet to be quantified. In this study, we used 4 years of surveys for wild and domestic ungulates to quantify the ecological success or failure of a single WMA by estimating wildlife and livestock densities before and after establishment, in comparison with a control site on adjacent land of similar size and habitat quality in the Tarangire Ecosystem, Tanzania. We defined ecological success in this case as significantly greater densities of wildlife and significantly lower densities of livestock in the WMA relative to the control site, after the WMA was established. Our study design was before-after-control-impact (BACI) to provide a strong basis for statistical inference (Green 1979; Stewart-Oaten et al. 1986; Underwood 1992). We defined the "impact" in this case as the establishment of the WMA. Inherent in the BACI design is the demonstration of similarity between impact and control sites during the "before" period, thus we predicted ungulate densities at impact and control sites would be similar before the WMA was established. We predicted higher resident wildlife densities and lower livestock densities in the WMA relative to the adjacent control site after the WMA was established, and that would indicate ecological success. Alternatively, if we found no detectable difference between the WMA and the control site, then we could conclude the WMA had no ecological effect during the course of our study.

Materials and Methods

We conducted 24 distance sampling surveys for wild and domestic ungulates from January 2012 to October 2015 in Randilen WMA (300 km^2) and in an adjacent control site in Lolkisale Game Controlled Area (LGCA; 200 km^2). Prior to WMA establishment, both sites were included in the LGCA and land-use management was similar under the authority of the Wildlife Division. In May 2014, Randilen WMA was established and new management activities were initiated, consisting mainly of ranger patrols intended to reduce poaching of wildlife and to reduce livestock and pastoralist presence in the WMA. Our control site in LGCA is an area adjacent to Randilen WMA that we used as a comparison site because rainfall, access to water, vegetation communities, land use, and topography are similar between the 2 sites, but the LGCA control site had few to no ranger patrols. WMA management allows pastoralist use of the WMA for grazing during times of drought, but no such drought occurred during the study period.

We estimated annual ungulate densities for each species in both sites using program DISTANCE 6.0 (Thomas et al. 2010). We first tested for between-sites similarity in annual densities during the "before" period of 2012 and 2013 using a 2-sample *z*-test for comparing 2 means (Buckland et al. 2001, 2015). We then tested for significant between-site differences in density in 2015, "after" management began in the Randilen WMA. Our criteria for ecological success were significantly greater ungulate wildlife densities and decreased livestock densities in Randilen WMA relative to LGCA in 2015, the first full year of surveys after the beginning of management in Randilen WMA.

Species surveyed were: cattle (*Bos taurus*), sheep (*Ovis aries*) and goats (*Capra hircus*) counted together, Kirk's dik-dik (*Madoqua kirkii*), steenbok (*Raphicerus campestris*), impala (*Aepyceros melampus*), plains zebra (*Equus quagga*), white-bearded wildebeest (*Connochaetes taurinus*), common waterbuck (*Kobus ellipsiprymnus*), and giraffe (*Giraffa camelopardalis*). All the non-domestic ungulates surveyed are categorized as Least Concern on the IUCN Red List, except the zebra, which is categorized as Near Threatened, and giraffe, which is Vulnerable (IUCN 2017).

Before-after-control-impact sampling is widely used in investigations of environmental impacts on abundance or density of a population. The principle is that an anthropogenic "impact" location will experience a different pattern of change from before to after the impact, compared with natural change in the control location (Underwood 1992). We assert that density is an appropriate criterion for assessing the performance of WMAs, particularly when nearby unmanaged areas or nearby protected areas of similar vegetation and habitat types are assessed simultaneously for comparison. Because we only had 1 impact and 1 control site, our inference is limited to the areas surveyed, but still is useful as a case study using formal impact assessment to quantify the ecological effectiveness of a WMA.

We surveyed according to a robust design sampling framework (Pollock 1982) with 3 sampling occasions per year near the end of each precipitation season (February, June, and October). Each sampling occasion was composed of 2 back-to-back sampling events during which we drove a fixed-route transect on dirt tracks in the study area, for a total of 6 sampling events per year. Driving speed was maintained between 15 and 20 kph on all transect routes, and all survey teams included the same 2 dedicated observers and a driver. Each track segment was sampled only 1 time in a given event. Each sampling event included a total transect length of 31 km in RWMA, and 48 km in LGCA. Transects were not randomly located but were systematic and representative of the 2 sites, and remained the same throughout the study; therefore, they are appropriate for making comparisons.

We collected distance data for all ungulates visible along both sides of the track out to 500 m. Distance data record the group size and perpendicular distance from the transect to each group of animals when first detected. When a group or singleton was sighted (groups for cattle, sheep and goats, dik-diks, steenbok, and impalas were defined as < 50 m between individuals, groups for zebras and wildebeest were defined as < 300 m between individuals, groups for giraffes were defined as < 400 m between individuals), we halted the vehicle and recorded the perpendicular distance from the track to the animal(s) measured with a laser rangefinder (Bushnell Arc 1000; Bushnell Outdoor Products, Overland Park, Kansas), the total number of individuals, and the GPS position of the vehicle. If the sighting was a cluster of animals, distance was measured as the perpendicular distance from the track to the middle of the group. Wildlife distances were recorded to the nearest meter, livestock data were binned into distances of 0–50, 51–100, 101–200, 201–300, 301–400, and 401–500 m. Our study design implemented identical replicate surveys to conduct an impact assessment by obtaining annual estimates of density for each site and calculating before-after similarity and difference between sites. Buckland et al. (2001, 2015) recommend systematic random designs, ≥ 10 replicate transect lines, and ≥ 60 observations for estimation. Our design did not conform to these recommendations, but our design of 6 surveys per year, where each survey was a single long transect in each site, was an appropriate design for our purposes.

Distance data for each species and site were analyzed separately with program DISTANCE 6.0 to estimate density of animals in each site while accounting for variation in detectability according to distance from the transect. We analyzed distance data following recommendations in Buckland et al. (2001, 2015). When comparing different sites, it is important to fit detection functions independently at each site. We analyzed all tracks surveyed within a site during a single sampling event as a single transect, and each of the 6 annual survey events were treated as replicate samples. We discarded the farthest 15% of observations. We plotted frequency histograms of perpendicular distances and fitted models to the histogram based on the key function and series expansion approach. We fit uniform, half-normal, and hazard-rate key functions with cosine, hermite, and simple polynomial series expansions. We fit the key function models and associated series expansions to the data and used the corrected Akaike information criterion (AICc) to select the best detection function model. We assessed goodness-of-fit of the top model using chi-square and Cramer von Misses tests. We regressed the logarithm of cluster size against the detection probability and adjusted detectability based on the expected cluster size.

We estimated year- and site-specific density using the top-ranked model for each site. Annual densities were computed using post-stratification. We tested whether our control and impact sites were similar during the "before" period (2012 and 2013) using *z*-tests of annual density estimates (Buckland et al. 2001, 2015). Confirmation of "before" similarity was established if annual density estimates were similar between sites before WMA management began. After WMA management began, we tested for ecological success using *z*-tests of annual density estimates in 2015 to determine if wildlife densities were significantly higher and livestock densities were significantly lower in the impact site (RWMA) relative to the control site (LGCA). If we found no detectable difference between WMA and unprotected land, then we could conclude the WMA had no ecological effect on monitored mammal species. The year 2014 was considered a transition year as one-half of the year was before and one-half was after the WMA management impact began, so we present the estimates but did not assign the year to before or after periods.

Results

We collected sufficient distance sampling observations for analyses of cattle, sheep and goats, impalas, giraffes, dik-diks, and zebras. Sufficient in our case was > 0 observations of the species in both sites in every survey. Summary statistics from DISTANCE analysis for density calculations, including figures of fitted detection functions, are available in Supplementary Data SD1. All detection functions passed goodness-of-fit tests, so we relied upon AICc to select the best function. The top-ranked detection functions were used to estimate annual densities for each species in each site.

Before the establishment of the WMA (years 2012 and 2013), annual densities of livestock and wildlife species were similar in control (LGCA) and impact (Randilen WMA) sites (Table 1; Fig. 1), indicating that our BACI study design was appropriate to discern the effects of WMA protections and management. In 2015, after implementation of WMA management activities, we found significantly higher densities of 2 species of resident wildlife (dik-dik and giraffe) and significantly lower density of cattle in Randilen WMA relative to LGCA (Table 1). Densities of cattle were highest inside Randilen WMA in 2014 due to a short-term invasion of cattle from outside the area. By 2015, the situation had reversed and density of cattle was lower inside the WMA. Although the difference did not reach statistical significance, a trend toward lower densities of sheep and goats in the WMA also was in the predicted direction (Table 1). One resident ungulate (impala) and 1 migratory species (zebra) showed similarities in density over time between Randilen WMA and LGCA in the before period, but no difference was detected in 2015 after the establishment and management of Randilen WMA (Table 1; Fig. 1). All surveyed species were observed in both sites before and after the establishment of the WMA, so no change in ungulate community composition was evident.

Table 1.—Summary statistics for annual differences in ungulate densities between Randilen Wildlife Management Area (RWMA) and adjacent Lolkisale Game Controlled Area (LGCA), Tanzania, before (2012 and 2013), during (2014), and after (2015) WMA establishment. Difference = RWMA − LGCA; *SE* for the diff. = $\sqrt{SE^2_{\mathrm{RWMA}} + SE^2_{\mathrm{LGCA}}}$; *z*-score = difference/*SE* for the diff.

	Before		During	After
	2012	2013	2014	2015
Cattle				
Difference	−0.36	−0.56	27.12	−19.07
SE for the diff.	9.83	5.95	16.77	9.31
z-score	−0.04	−0.09	1.62	−2.05
P-value	0.49	0.46	0.05	0.02
Sheep and goats				
Difference	7.26	9.17	3.20	−33.94
SE for the diff.	10.63	5.78	4.36	27.72
z-score	0.68	1.59	0.73	−1.22
P-value	0.25	0.06	0.23	0.11
Giraffe				
Difference	0.02	0.09	0.24	0.54
SE for the diff.	0.22	0.18	0.24	0.31
z-score	0.10	0.51	1.03	1.76
P-value	0.46	0.31	0.15	0.04
Dik-dik				
Difference	0.19	−0.19	−0.16	0.80
SE for the diff.	0.26	0.54	0.58	0.25
z-score	0.72	−0.35	−0.27	3.12
P-value	0.24	0.36	0.39	< 0.01
Impala				
Difference	−2.52	−0.60	−1.73	−4.25
SE for the diff.	1.59	1.86	3.04	4.00
z-score	−1.59	−0.32	−0.57	−1.06
P-value	0.06	0.37	0.28	0.14
Zebra				
Difference	3.49	−1.86	3.57	0.92
SE for the diff.	4.11	2.97	9.78	0.97
z-score	0.85	−0.63	0.36	0.95
P-value	0.20	0.26	0.36	0.17

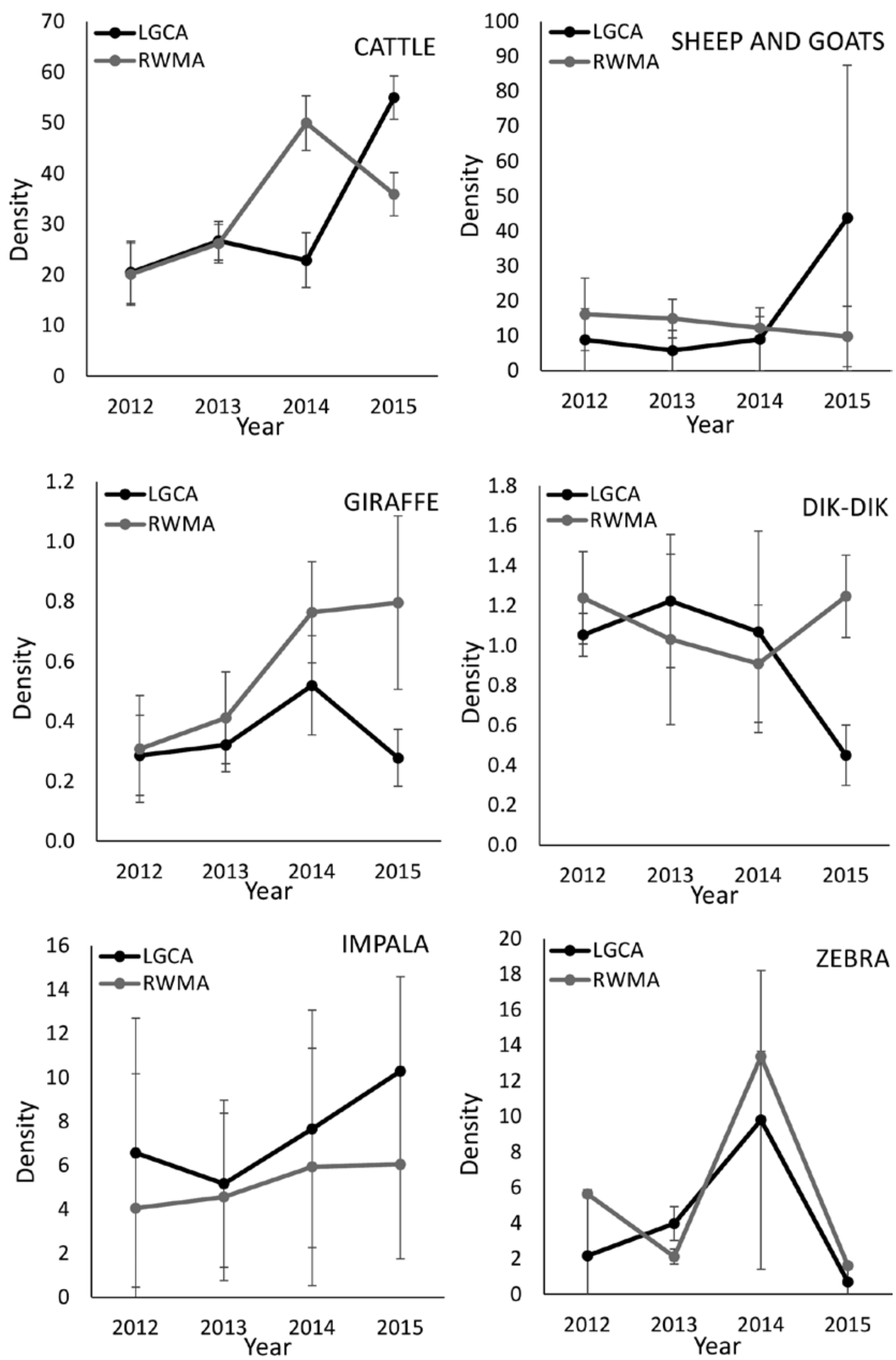

Fig. 1.—Mean annual densities (#/km^2, ± *SE*) of livestock and wildlife species in Randilen Wildlife Management Area (RWMA = impact area) and adjacent Lolkisale Game Controlled Area (LGCA = control area), Tanzania, from 2012 to 2015. Randilen WMA was established and management activities began in May 2014, making 2014 the "establishment" period, 2012 and 2013 are the "before" period, and 2015 is the "after" period in the before-after-control-impact design.

Discussion

Since their inception, the performance of community-based conservation efforts has been debated, and policies promoting CBNRM have faced opposition (Naughton-Treves et al. 2005; Roe 2008). The success record of CBNRM is mixed, with many documented ecological and economic failures (Kellert et al. 2000; Blaikie 2006; Singleton 2009; Brooks et al. 2012; Measham and Lumbasi 2013; Salerno et al. 2015). However, quantitative evidence suggests positive social and ecological outcomes can also result from CBNRM projects (Tallis et al. 2008; Brooks et al. 2012). Our data demonstrate that WMA establishment and management in Tanzania as practiced in Randilen WMA had positive ecological outcomes for some resident mammalian wildlife species over the short-term period of this study. All significant results after RWMA became operational conformed to our predictions for ecological success, namely higher wildlife densities and lower livestock densities in the WMA relative to the control site. Our estimates of wildlife and livestock densities were similar to other estimates from the region (Kiffner et al. 2016), with the exception that cattle densities were much lower in our areas relative to those reported from only 20 km away to the northwest (Kiffner et al. 2016).

The rapid change in densities of resident wildlife following WMA establishment could be due to several mechanisms, but is most likely the result of a change in spatial distribution following the shifted distribution of livestock and pastoralists and implementation of protective anti-poaching patrols in Randilen WMA. Giraffes may have shifted their distribution into the WMA as a result of lower relative density of humans and livestock inside the WMA. It also is possible that habitat change such as increased cover because of fewer livestock or reduced poaching pressure may have mediated the difference we documented for dik-diks. Despite potential negative effects of livestock on wildlife density, wildlife species are able to coexist with livestock at relatively high densities given protection from illegal hunting and increasing people's tolerance of wildlife (Kinnaird and O'Brien 2012). Regardless of the mechanisms involved, the apparent positive ecological effects we found in Randilen WMA should provide evidence that CBNRM in Tanzania via WMA establishment likely compliments the conservation value of national parks (Leménager et al. 2014), especially for smaller parks like Tarangire (Borner 1985; Prins 1987). Here, our impact assessment included only 1 control and 1 impact site monitored over a relatively short time frame; therefore, we suggest similar evaluation efforts be conducted in multiple WMAs to confirm our results and expand upon our findings.

Most CBNRM programs are aimed at promoting conservation while maintaining or improving people's standards of living. Our study did not assess impacts of RWMA establishment on local human livelihoods, and that aspect of CBNRM should be formally quantified for WMAs. Most CBNRM programs have only limited success at achieving both conservation and human development goals, and although linking conservation with development may be desirable, the simultaneous achievement of these 2 objectives may be impossible because of inherent contradictions (Newmark and Hough 2000). Despite the apparent positive ecological outcome we detected, our results do not imply that current efforts are sufficient to sustain this WMA's ecological success in the longer term. Populations of livestock and resident ungulates should be continuously assessed, so management actions can be evaluated. Assessing the variability of population densities of wildlife species over time requires regular long-term monitoring (Link et al. 1994), and monitoring within WMAs should always be in reference to nearby areas of similar vegetation and habitat types. Ideally, locally based monitoring schemes should replace foreign scientists, as locally based monitoring has the potential to reinforce community-led resource management systems and lead to more sustainable wildlife conservation (Danielsen et al. 2005).

Supplementary Data

Supplementary Data are available at *Journal of Mammalogy* online.

Supplementary Data SD1.—Summary statistics from DISTANCE analysis for densities of large mammals surveyed in Randilen Wildlife Management Area and Lolkisale Game Controlled Area, Tanzania.

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