

Effect of Diversity of Large Wildlife Species on Financial Benefits to Local Communities in Northwest Namibia

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Accepted: 13 May 2010 / Published online: 22 September 2010
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Abstract There exist few quantitative assessments of the relationship between biodiversity *per se* and economic benefits at scales that are relevant for conservation. Similarly, the merits of Community-Based Natural Resource Management programs for both wildlife and people are contested. Here, we harness two databases, on wildlife surveys and financial benefits, to address these issues for communal conservancies in northwest Namibia. We use ordination methods to characterize the diversity and stability of large wildlife assemblages on conservancies, and demonstrate that diversity (but not stability) is an important explanator of conservancy financial benefits. Our results indicate that for this area of Namibia, biodiversity, as represented by large wildlife assemblages, has an important, positive effect on the tangible financial benefits that people derive from conservation programs.

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Keywords Africa · Biodiversity · Community-based conservation · Community ecology · Ecosystem services · Diversity · Ecotourism · Stability · Wildlife

1 Introduction

Community-Based Natural Resource Management (CBNRM) is increasingly discussed as a way of improving human welfare while conserving (or restoring) wildlife and other elements of biodiversity (Adams et al. 2004; Berkes 2004). Local exploitation of the resource base is seen as fundamental to sustainable development strategies in much of southern Africa, and devolution of resource management to local institutions has been adopted as a key policy reform by several countries in this region, among others (Murombedzi 1999; Barnes et al. 2002). Empirical evaluation of the effects of this approach on both biodiversity and people is necessary (Stuart-Hill et al. 2005), as is an improved understanding of the relationship between biodiversity conservation and human well being. Nevertheless, rigorous evaluations of the impact of most types of conservation interventions are strikingly scarce (Ferraro and Pattanayak 2006), not only in southern Africa, but everywhere conservation is practiced (but see Wilkie et al. 2006).

At experimental or mesocosm scales, numerous studies have shown that more diverse systems with higher levels of metrics such as stability and resistance lead to greater performance of ecological systems (Balvanera et al. 2006; Cardinale et al. 2006; Tilman et al. 2006). These results are often used as justification for the conservation of biodiversity in natural systems. However, there are few studies that have explicitly quantified biodiversity and explored its relationship to the economic benefits that people derive from nature at scales relevant for both conservation and development (Srivastava and Vellend 2005). Yet real-world examples of what has been demonstrated in the laboratory are critical if policy-makers and others are to be convinced of the material benefits of conserving biodiversity.

Wildlife ecology research in southern Africa is primarily conducted within game management, range management, and/or population ecology frameworks (Du Toit and Cumming 1999; Du Toit 2002; Grange and Duncan 2006). As such, community ecology perspectives are under-represented (though see Gardner et al. 2007 for an example from East Africa) and could provide complementary insights into how ecological systems can be managed to provide both biodiversity and economic benefits to people. Community ecology methods can characterize wildlife diversity using ordination methods and metrics such as stability, resilience, and resistance, and are an important means of quantifying biodiversity across space and time (Legendre and Legendre 1998). Given the emphasis on community ecology methods in the experimental literature, it appears timely to investigate how such techniques could lead to improved understandings of the relationships between biodiversity and human well-being in southern Africa.

We describe here an analysis that aims to explore the linkages between the diversity of large wildlife assemblages and the economic success of CBNRM management institutions in northwest Namibia. Harnessing separate databases on financial benefits and wildlife surveys, we use ordination methods to characterize the diversity of large wildlife on conservancies, and multiple regression analysis to assess the relationship between economic benefits and wildlife diversity. Our results illustrate that the diversity of large wildlife species positively affects financial returns in a CBNRM program in Namibia. They also demonstrate the potential for using community ecology analytical methods to better understand relationships between biodiversity and economic benefits in real-world conservation applications.

2 Methods

2.1 Study Site

Our study area is the northwest region of Namibia (Fig. 1). The area is characterized by low rainfall (80–300 mm), rugged and mountainous terrain (mean elevation \sim 900 m, standard deviation \sim 200 m), and relatively low levels of anthropogenic land use. Vegetation ranges from sparse shrubs and grasses in the western desert region to a mixture of grasslands, trees, and shrubs in the wetter eastern portion (Mendelsohn et al. 2002). Despite the aridity, the region also houses a relatively diverse set of wildlife species (NACSO 2008), and the Namib desert has been identified as an area of global conservation importance due to high levels of plant endemism (Olson and Dinerstein 1998). Whilst much of the study area is too dry for permanent settlement, humans have nevertheless been present on the landscape for millennia (Mendelsohn et al. 2002). The original human populations in the area were hunter-gatherers and nomadic pastoralists, but this has now given way to a sedentary lifestyle involving livestock farming in combination with small gardens. With the advent of the conservancy system, a variable but increasing fraction of communal lands is being managed primarily for wildlife—from 12 to 100% in a sample of 5 conservancies, ca. 2000 (Barnes et al. 2002).

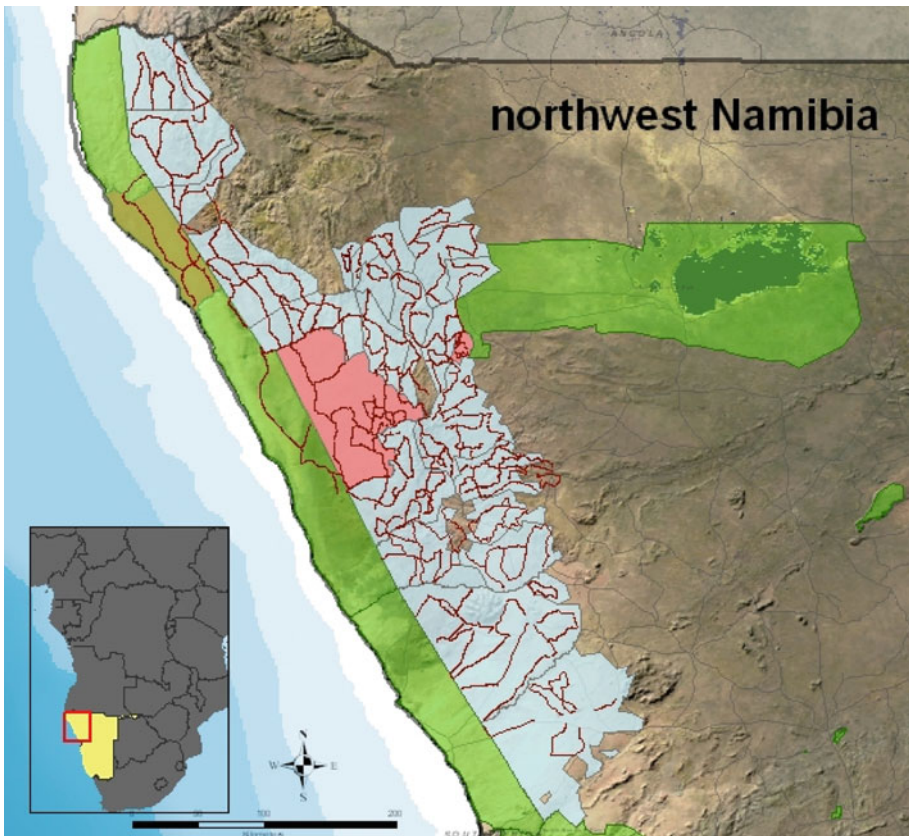


Fig. 1 Northwest Namibia, with locations of wildlife count transects (*lines*) on communal conservancies (*blue*), concessions (*red*), and protected areas (*green*)

2.2 Conservancies

In Namibia, as in much of southern Africa, a colonial regime and extended armed conflict (1966–1989), along with increasing human densities, competition with livestock, and drought resulted in precipitous declines of wildlife populations until the mid 1990s (NACSO 2008). In response, the Namibian government enacted progressive legislation that devolved conditional user rights over natural resources such as wildlife to private landholders, and eventually, to communities on customary landholdings. Since 1998 communities have been able to register their communal lands as “conservancies”; to date about 39% of all communal lands in Namibia have been registered as such. Income to conservancies has grown at an increasing rate since 1998, and wildlife populations in most of Namibia have been increasing since the 1980s (NACSO 2008), but there has been little quantitative research on the factors that affect variation in financial benefits across conservancies in the northwest region where annual wildlife monitoring occurs.

Being designated as a conservancy allows local communities to profit from game hunting and tourism on their communal lands. In exchange, conservancies must develop and implement management plans, which include (i) a higher level goal accompanied by a set of strategies to achieve this goal; (ii) a ‘zonation map’ which is a vision for how competing forms of land use in the conservancy can be separated spatially; (iii) an annual work calendar that specifies what should be done, when and by whom in order to ensure that the annual work cycle of the conservancy is completed; and (iv) a development plan that articulates the longer-term development vision of the conservancy. A key part of the management plan is proposing sustainable off-take rates of wildlife species; off-take rates are debated and proposed by conservancy membership, incorporate annual monitoring of wildlife and resultant population estimates, and are reviewed by the Ministry of Environment annually. Off-take rates for common species in the northwest average from 0.5 to 8% of the minimum estimate of population size on a conservancy. Failure to comply with agreed upon off-take levels can result in the suspension of hunting quotas.

2.3 Wildlife Assemblages

Since 2000, visual surveys of large vertebrates (>5 kg) have been conducted by driving permanent transects throughout the northwest region of Namibia during the month of June (Fig. 1). Relative abundances (individuals/100 km) of all observed species were determined across 18 conservancies and 4 additional management units (either protected areas or tourism/hunting concessions) for each year in which transects were driven. We hereafter refer to all 22 units as conservancies, except where noted. Not all conservancies had observations extending back to 2000, though all had at least 3 years of records. Ad-hoc or casual observations were used to supplement instances where species were known to be present on conservancy lands, but were not recorded on transects. In these cases, a small positive value was added to the species-by-site matrix in order to prevent misleading effects of false “double-zeros” in the ordination analysis described below (Legendre and Legendre 1998). We also excluded Black-faced Impala (*Aepyceros melampus petersi*) from our analyses because the only observation was of a large herd in one conservancy that unduly skewed ordination results.

Our wildlife dataset therefore consisted of abundance estimates at 22 sites for 20 species¹ over multiple years. Treating each site-year combination as a separate observation,

¹ The species considered in this analysis are Bat-eared fox (*Otocyon megalotis*), Black-backed jackal (*Canis mesomelas*), Black rhino (*Diceros bicornis*), Bush duiker (*Sylvicapra grimmia*), Chacma baboon

our data matrix had 172 rows and 20 columns. We used a multivariate technique from the community ecology literature, Nonmetric Multidimensional Scaling (NMDS), to characterize conservancies according to their wildlife diversity (Legendre and Legendre 1998). NMDS is an ordination technique that, like correspondence analysis and principal components analysis, seeks to boil down multidimensional relationships among data to a smaller subset of dimensions that captures the dominant gradients and is easier to visualize and interpret (Legendre and Legendre 1998). In contrast to the above mentioned ordination techniques, NMDS makes no assumptions regarding the shape of the functional relationship between species abundances and underlying environmental gradients.

The first step of a NMDS is transforming the original site-by-species matrix into a matrix which contains values that reflect the (dis)similarities in the large wildlife assemblage between each pair of sites. For each pair of sites, a coefficient is calculated that reflects the distance between the two sites using the original large wildlife abundance estimates. We chose a commonly-used distance coefficient, the Bray-Curtis coefficient, that is appropriate for abundance-based species estimates (Legendre and Legendre 1998). For each pair of sites (a , b), the value:

$$D_{ab} = 2W/(A + B) \quad (1)$$

where A is the sum of abundances, over all species, at site a , B is the sum of species abundances at site b , and W is the sum over all species of [Min(abundance of species k at site a , abundance of species k at site b)].

The iterative NMDS algorithm then takes the distance matrix and plots objects to minimize the differences among points between the original distance matrix and the new position of points along (typically) 2 or 3 axes. The algorithm is iterative in that a particular configuration is tried, a matrix of fitted distances is calculated, these fitted distances are regressed on the original distances, and a goodness of fit measure is calculated. Numerical optimization techniques are used to iteratively improve on this goodness of fit measure (called stress) until convergence or a pre-identified tolerance in lack of fit is reached.

An initial plot of stress (the measure of the difference between original and scaled points referred to above) versus dimension showed that NMDS with dimensions greater than 1 all fell within the acceptable 10–20 limit that is typical of ecological applications (McCune and Grace 2002). Restricting detailed examination to the two and three dimension solutions, we found that environmental variables did not correlate strongly with a third axis, and therefore, chose a two-dimensional solution to represent diversity of large wildlife on conservancies.² To interpret the resulting ordination space, we created biplots (Legendre and Legendre 1998) for component species abundances against site-years, and for environmental

Footnote 1 continued

(*Papio ursinus*), Cheetah (*Acinonyx jubatus*), Common warthog (*Phacochoerus africanus*), Damara dik-dik (*Madoqua kirkii damarensis*), Eland (*Taurotragus oryx*), Elephant (*Loxodonta africana*), Gemsbok (*Oryx gazelle*), Giraffe (*Giraffa camelopardus*), Greater kudu (*Tragelaphus strepsiceros*), Hartmann's mountain zebra (*Equus zebra hartmanni*), Klipspringer (*Oreotragus oreotragus*), Leopard (*Panthera pardus*), Lion (*Panthera leo*), Ostrich (*Struthio camelus*), Springbok (*Antidorcas marsupialis*), and Steenbok (*Raphicerus campestris*).

² In addition to the analyses that we present here, we also performed additional NMDS ordinations to assess how sensitive our results were to less-common species. We conducted NMDS on (i) a species-site abundance matrix that only included the most commonly recorded species (Black-backed jackal, Gemsbok, Giraffe, Greater kudu, Hartmann's mountain zebra, Klipspringer, Ostrich, Springbok, and Steenbok), and (ii) a presence-absence only matrix of all species. Results from the common-species data matrix were very similar to those we present here; the species most strongly correlated with axis one and two were the same in both analyses, and the axes from both ordinations were highly correlated with one another (axis 1: $r = 0.92$; axis 2: $r = 0.75$). In addition, we also varied the value of the small positive number (0.01 vs. 0.1 vs. 1) added

variables against site-years. Each point in the biplots represented the species assemblage for a given site in a given year. The distance between points is a representation of how similar sites are to one another, based on their large wildlife assemblages. The environmental variables that we hypothesized might affect the structure of wildlife assemblages were precipitation, variation in precipitation, elevation, variation in elevation, soil fertility, livestock density and conservancy area; these data were collated from a number of local databases (Mendelsohn et al. 2002; NACSO 2008).³ We also classified conservancies according to their dominant Global Land Cover 2000 land-use category (Bartholome and Belward 2005) and coded symbols accordingly.

Our measure of “stability” of large wildlife assemblages on conservancies was the degree of interannual variability in a conservancy’s position in NMDS space. For each conservancy, we calculated the average (x , y) NMDS coordinate over all years in which wildlife surveys took place. The distance from each of the individual yearly points to the average position was then calculated for each conservancy, and stability was estimated as the average of these distance vectors. To visualize stability, we plotted all conservancy-year points with lines that attach them to their average NMDS position.

2.4 Conservancy Financial Benefits

To assess the effect of the diversity and stability of wildlife assemblages on conservancy financial benefits, we used comprehensive financial data collected annually for all conservancies in Namibia (NACSO 2008). Average annual financial benefits (in Namibian dollars; 7.5 N\$~1 USD as of February 2008) were defined as the sum of conservancy cash income, employment income to conservancy members, and financial value of in-kind contributions such as harvested game meat, training to conservancy members, and other private sector contributions to the conservancy (Barnes et al. 2002). The majority (~80%) of financial benefits are from tourism and safari hunting operations, but conservancies vary in the actual mix of income they derive from hunting vs. ecotourism vs. other sources.

2.5 Modelling Approach

Many factors other than the diversity of large wildlife species will influence a conservancy’s financial benefits. Due to our limited sample size, however, we could not include every potentially relevant variable. Our approach therefore, was to place relevant variables into either a “biophysical” or “social” effects category, and select one proxy variable from each category that was highly correlated with others in its class. The biological variables we considered for conservancies were the mean and standard deviation of both elevation and precipitation,

Footnote 2 continued

to the data matrix for those species known to be present on conservancies but not recorded on formal transects; this did not change the resulting ordination (correlation coefficients >0.9 for all axes in all cases). On the other hand, the presence-absence matrix resulted in a different ordination; different species were correlated with axes 1 and 2 and the axes from the presence-absence NMDS were not well correlated with those that resulted from either of the abundance-based NMDS. We therefore present only the abundance-based NMDS results, as they appear robust to various changes in the underlying data matrix, and also give a richer picture of wildlife assemblages than one based only on species presence-absence.

³ We also assessed whether spatial effects, apart from the environmental variables we considered, were related to variation in large wildlife assemblages. We conducted a Mantel-test on the wildlife dissimilarity matrix and on a dissimilarity matrix based on the geographical distance between conservancies (as represented by the coordinates of their geographic centroid). The test statistic was not-significant (Mantel statistic r : -0.0076, p = 0.588), indicating that there were no “pure spatial” effects of a conservancy’s position in geographic space on their large wildlife assemblages (i.e., no spatial autocorrelation).

Table 1 Correlations among variables that describe the biophysical status of conservancies

	elev.avg	elev.std	prec.avg	prec.std	soil.fert	major. wildlife	big5	rhino	GLC.cat
elev.avg	1	-0.49	0.9161	-0.3147	0.0246	-0.0627	-0.1251	-0.383	0.9225
elev.std	-0.49	1	-0.6503	0.8122	0.2151	-0.0458	0.0308	0.328	-0.5588
prec.avg	0.9161	-0.6503	1	-0.3983	0.0714	0.1987	0.0615	-0.267	0.9141
prec.std	-0.3147	0.8122	-0.3983	1	0.2483	-0.0399	0.0505	0.339	-0.3904
soil.fert	0.0246	0.2151	0.0714	0.2483	1	0.3107	0.3568	0.266	0.08
major. wildlife	-0.0627	-0.0458	0.1987	-0.0399	0.3107	1	0.7907	0.704	0.0273
big5	-0.1251	0.0308	0.0615	0.0505	0.3568	0.7907	1	0.838	-0.0379
rhino	-0.3827	0.3283	-0.2667	0.3394	0.2661	0.7044	0.8381	1	-0.3254
GLC.cat	0.9225	-0.5588	0.9141	-0.3904	0.08	0.0273	-0.0379	-0.325	1

Variable descriptions: *elev.avg* Average elevation in metres, *elev.std* Standard deviation of elevation, in metres, *prec.avg* Average mm of precipitation, *prec.std* Standard deviation of precipitation, in mm, *soil.fert* Relative suitability of soil for crop cultivation, *major wildlife* Number of large wildlife species present on a conservancy, *big5* Number of “Big Five” species present on conservancy, *rhino* Presence/absence of black rhino (*Diceros bicornis*), *GLC.cat* Global Land Cover 2000 category (Bartholome and Belward 2005)

average soil fertility, the number of large mammal species present, the number of “Big 5” species present, and whether black rhino (*Diceros bicornis*; a particularly high value species for tourism) were present. Social variables considered were years since conservancy establishment, size of conservancy, distance from Namibia’s major tourist circuit, livestock densities, human population, number of ethnic groups present, size and gender balance of management committee, the number of support agencies that working with a conservancy, and the rating of the conservancy’s management capacity from an independent assessment (Stuart-Hill et al., unpublished data). Data sources were the same as for the wildlife analysis explanatory variables.

Correlation analysis among biophysical variables showed that average precipitation was strongly correlated with elevation and with land cover categorization (Table 1). In an arid region such as northwest Namibia, precipitation exerts a dominant effect on ecological processes, therefore we used this variable as a surrogate for major biophysical differences among conservancies. Among social variables, the number of years since conservancy establishment was correlated with the number of support agencies working with conservancies, the number of ethnic groups within a conservancy, and the assessment of conservancy management capacity (Table 2). Older conservancies have generally had greater levels of support, which has led to greater management capacity. In addition, there is probably a selection bias: those areas that were more pre-disposed towards success were probably the first to have become conservancies. We therefore chose years since establishment as a surrogate for social variables that might affect conservancy income.

We used an Ordinary Least Squares multiple regression approach to assess the effects of wildlife diversity and stability on conservancy benefits, while including precipitation and years since conservancy establishment as proxies for other important biological and social effects. Sample size for this analysis was 18, since the 4 non-conservancy management units did not have associated financial data. To improve normality of the response variable we log-transformed conservancy benefits. We standardized each of the potential explanatory variables by subtracting their mean and dividing by their standard deviation, to allow easier comparison of resulting regression coefficients. For all analyses we used the software package R (R Development Core Team 2008).

Table 2 Correlations among variables that describe social status of conservancies

	Year_reg	area.km ²	live.dens	d.tour.rte	pop	sum.NR	Ethnic	committee	comm.women	supp.agencies
Year_reg	1	-0.2182	0.328	0.2071	0.1557	-0.6093	-0.5496	0.206	-0.0406	-0.5988
area.km ²	-0.2182	1	-0.44	-0.2674	0.0202	0.5436	0.2569	0.0204	0.2178	0.0453
livestock.dens	0.3284	-0.44	1	0.1981	0.2588	-0.3552	-0.1305	0.117	-0.2874	-0.3528
dist.tour.route	0.2071	-0.2674	0.198	1	-0.5663	-0.0159	-0.1446	0.1492	-0.0174	-0.417
pop	0.1557	0.0202	0.259	-0.5663	1	0.0518	-0.0438	-0.1063	-0.0677	-0.1461
sum.NR	-0.6093	0.5436	-0.355	-0.0159	0.0518	1	0.563	0.0342	0.2504	0.0539
Ethnic	-0.5496	0.2569	-0.13	-0.1446	-0.0438	0.563	1	0.0734	0.1352	0.1322
committee	0.206	0.0204	0.117	0.1492	-0.1063	0.0342	0.0734	1	0.7908	-0.2884
committee.women	-0.0406	0.2178	-0.287	-0.0174	-0.0677	0.2504	0.1352	0.7908	1	-0.1043
support.agencies	-0.5988	0.0453	-0.353	-0.417	-0.1461	0.0539	0.1322	-0.2884	-0.1043	1

Variable descriptions: *Year_reg* number of years since the conservancy was registered, *area.km²* area, in square kilometers, *live.dens* stocking density of livestock (kg/ha), *d.tour.rte* distance, in km, to established northwest Namibia tourist circuit, *pop* population, *sum.NR* qualitative score on conservancy performance of natural resource management (Stuart-Hill et al., unpublished data), *Ethnic* number of ethnic groups living within borders, *committee* number of people sitting on the conservancy management committee, *comm.women* number of women on conservancy management committee, *supp.agencies* number of agencies that support management of the conservancy

3 Results

The two dimensional NMDS solution resulted in a stress of 14.83, within the 10–20 range that typifies acceptable applications in the ecological sciences (McCune and Grace 2002). Biplots of the position of conservancies along NMDS axes one and two (Fig. 2, upper panel) revealed several prominent environmental gradients. Running roughly along axis two, there was a strong gradient associated with both elevation and precipitation. Conservancies in the upper right hand corner of Fig. 2, upper panel, are conservancies at higher elevations with higher rainfall, and with land cover characterized either by open grasslands, or by savannas (grasslands with sparse tree and shrub cover). Along a more horizontal plane in the figure, conservancies to the right-hand side were those with higher livestock densities and savanna vegetation. Conservancies on the lefthand side had higher soil fertilities, but were mostly desert vegetation.

The ordination space can also be interpreted in terms of the underlying species abundance matrix (Fig. 2, lower panel). Conservancies on the righthand side of the figure are relatively depauperate areas containing steenbok (*Raphicerus campestris*) but little else, probably relating to an inability of wildlife to compete with higher livestock and human densities. Abundances of most species are highest on the lefthand side of the figure. In particular, ordination space is strongly correlated with ostrich (*Struthio camelus*), gemsbok (*Gazella oryx*), Hartmann's mountain zebra (*Equus zebra hartmannae*), and kudu (*Tragelaphus strepsiceros*), with the two former species more abundant in lower elevation desert areas, and mountain zebra and kudu found at higher elevations on grasslands.

A conservancy's position in wildlife ordination space varied across years (Fig. 3). Some conservancies had relatively stable wildlife assemblages from year to year (e.g., Gaingu, Tsiseb), whereas others fluctuated considerably (e.g., Anabeb, Sanitatas). Correlation analysis of this annual variation in ordination scores showed that it was unrelated to biophysical features such as position along NMDS axis 1 ($r = 0.066$) or axis 2 ($r = 0.30$), or precipitation ($r = 0.45$). Instead, the majority of the variation was explained by the size of the conservancy ($r = -0.82$), indicating that smaller conservancies had more variability in their annual wildlife diversity scores than larger ones. We corrected for this area effect by taking the residuals from a regression between stability and area, using this as our measure of the stability of large wildlife assemblages in subsequent analyses.

The strongest parameters in the model of (log-transformed) per-capita conservancy benefits were NMDS axis one (negative effect) and the conservancy's age (positive effect; Table 3). NMDS axis two also had a significant negative effect on conservancy benefits. The coefficients for precipitation and temporal stability of wildlife assemblages were not significantly different from zero. This indicates that holding a conservancy's age constant, those with lower scores for both NMDS axis 1 and NMDS axis 2 (i.e., those conservancies in the lower lefthand part of the ordination space) are expected to have greater per-capita benefits. The R^2 for the model was 0.909, indicating that about 91% of the variance in conservancy benefits was explained by our statistical model (Fig. 4).

4 Discussion

Many in the conservation community hold the view that systems with high levels of biodiversity are likely to be the most economically valuable, but this has rarely been demonstrated in natural settings and at scales relevant to real-world conservation (Srivastava and Vellend 2005). Here we have shown that after controlling for other confounding variables,

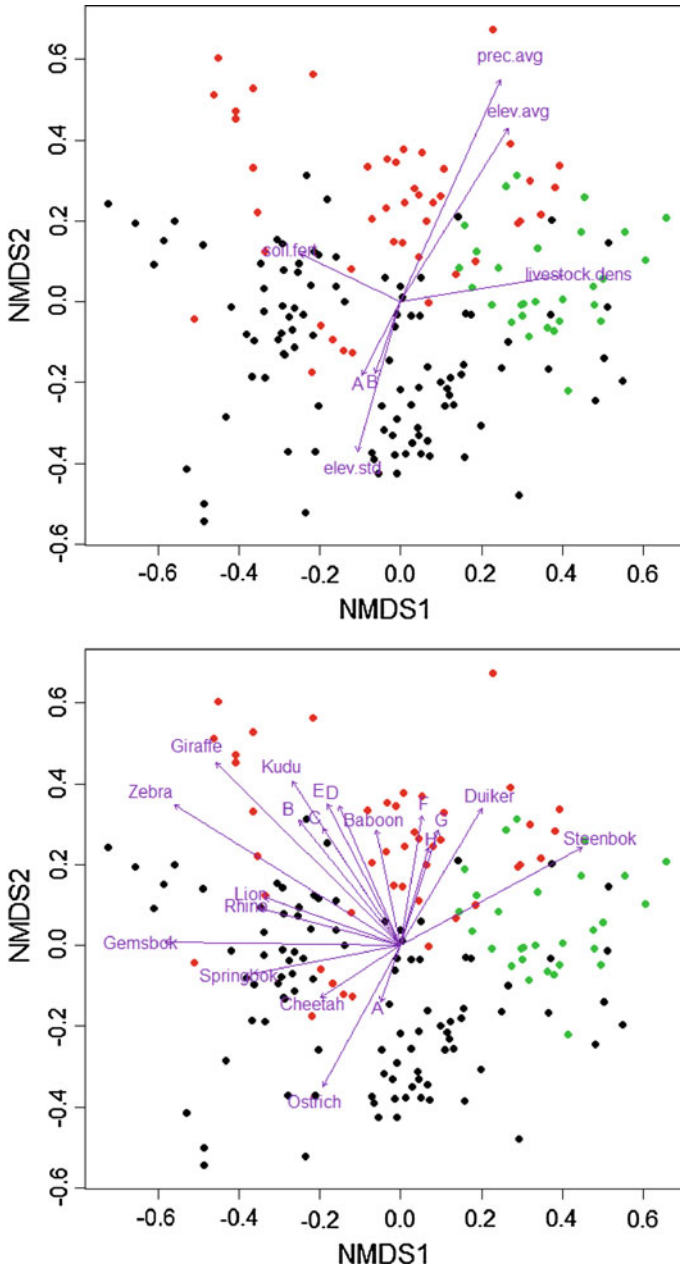


Fig. 2 Two-dimensional biplot representation of wildlife community structure in northwest Namibia. Points represent conservancy-years. Colours represent dominant land cover types; *red* open grasslands, *green* savanna (grassland with shrubs/trees), *black* desert. Environmental gradients (*upper panel*) and species vectors (*lower panel*) are also shown. Upper panel: *A* area.km², *B* prec.std. Lower panel: *A* Bat-eared fox, *B* Jackal, *C* Elephant, *D* Warthog, *E* Eland, *F* Klipspringer, *G* Dik-dik, *H* Leopard

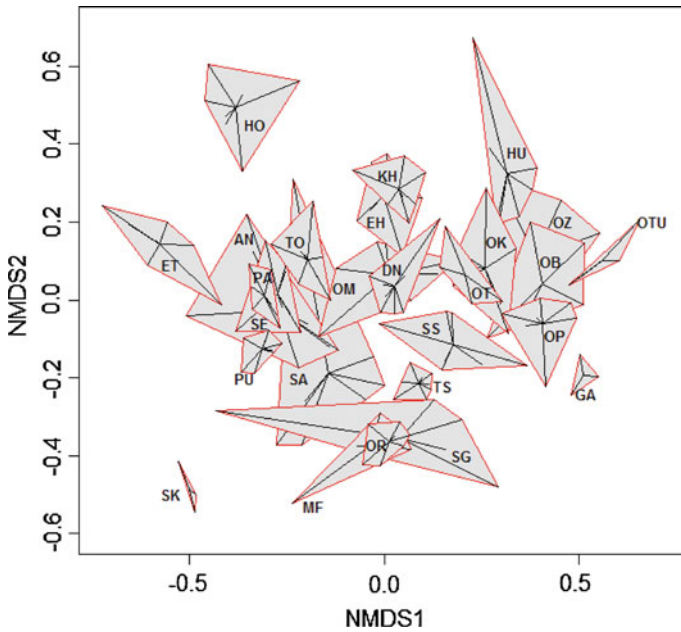


Fig. 3 “Stability” of wildlife assemblages on conservancies [Abbreviations (registered conservancy unless otherwise noted): *AN* Anabeb, *DN* Doro !Nawas, *EH* Ehirovipuka, *ET* Etendeka concession, *GA* #Gaingu, *HO* Hobatere concession, *HU* //Huab, *KH* #Khoadi //Hoas, *MF* Marienfluss, *OK* Okangundumba, *OM* Omantendeka, *OP* Orupupa (not yet registered), *OR* Orupembe, *OT* Otjambangu, *OB* Otjimboyo, *OTU* Otuzemba (not yet registered), *OZ* Ozondundu, *PA* Palmwag concession, *PU* Puros, *SA* Sanitatas, *SG* area between Sanitatas and Puros conservancies, *SE* Sesfontein, *SK* Skeleton Coast national park, *SS* Sorris Sorris, *TO* Torra, *TS* Tsizeb.] in northwest Namibia: length of line from center equals the distance of an individual year score from the average ordination score. Minimum convex polygons are for illustration clarity only

Table 3 Coefficients for regression model of log-transformed per-capita benefits

	Estimate	Std. Error	<i>t</i> -value	<i>P</i>
(Intercept)	−5.471	1.519	−3.602	0.00363
ln.points1	−4.309	0.962	−4.481	0.00075
ln.points2	−3.300	1.305	−2.529	0.02645
ln.years.old	5.132	0.819	6.266	0.00004
ln.stab.res	0.066	0.813	0.081	0.937
prec.avg	0.034	0.291	0.118	0.908

more diverse wildlife assemblages on conservancies in Namibia have a strong positive effect on conservancy financial benefits. While temporal variability in wildlife diversity was not important, the average scores along both axes of an ordination were important explanators of conservancy income.

How do we interpret these findings? Wildlife on conservancies in Namibia are an important aspect of a conservancy’s natural wealth, from a consumptive standpoint (animals are hunted both for trophies and for their meat and hides), and from a tourism standpoint (large vertebrates, especially charismatic species such as lions, elephants, and rhinos, are major draws for tourists). Our results go further and suggest that the structure and diversity of wildlife assemblages is in fact a dominant factor predicting conservancy financial benefits. Conservancies

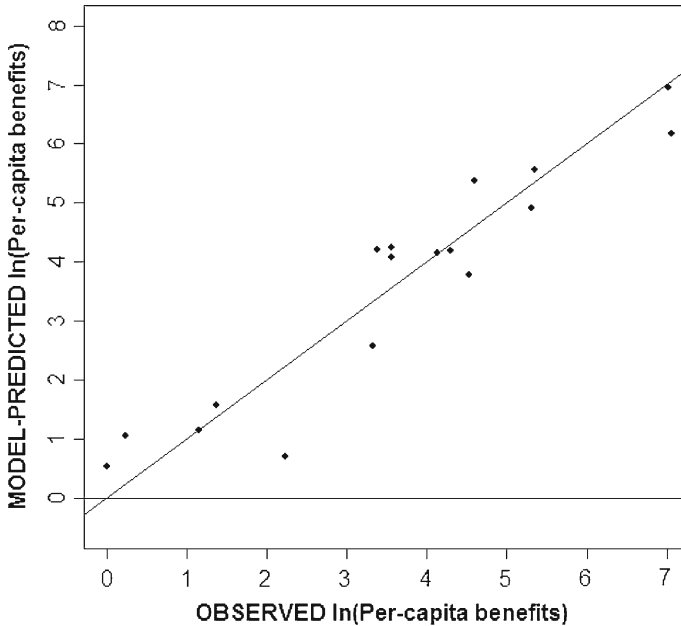


Fig. 4 Model-predicted vs. observed values of log-transformed per-capita conservancy benefits

with a diverse assemblage of wildlife (i.e., those on the lefthand side of Fig. 2) had higher per-capita benefits than conservancies with depauperate wildlife assemblages, even after we controlled for biophysical and social variables that should also regulate benefit levels. More complex and diverse wildlife assemblages allow for greater hunting options, and are also likely to provide tourists with a greater range of viewing opportunities.

We are unaware of other studies that have explicitly linked the diversity of wildlife assemblages (as characterized from a community ecology perspective) to the economic gains of local peoples. In Namibia, the great majority of conservancy financial benefits are due to income generated from trophy hunting or photographic tourism. Recent reviews of trophy hunting from Namibia and other countries in southern Africa show convincingly that hunting can be a profitable land use and can generate considerable benefits to local and national economies, but these studies do not present data on the relationship between economic gains and the structure of wildlife assemblages (Humavindu and Barnes 2003; Lindsey et al. 2007; Samuelsson and Stage 2007). Similarly, tourism can also generate considerable income and be a positive force for conservation (Walpole and Leader-Williams 2001; Kruger 2005), but quantitative information on the relationship between wildlife or biodiversity and revenues from tourists is generally lacking (Naidoo and Adamowicz 2005). In this context, our results provide tangible evidence that biodiversity per se can positively influence economic returns to lands under conservation management.

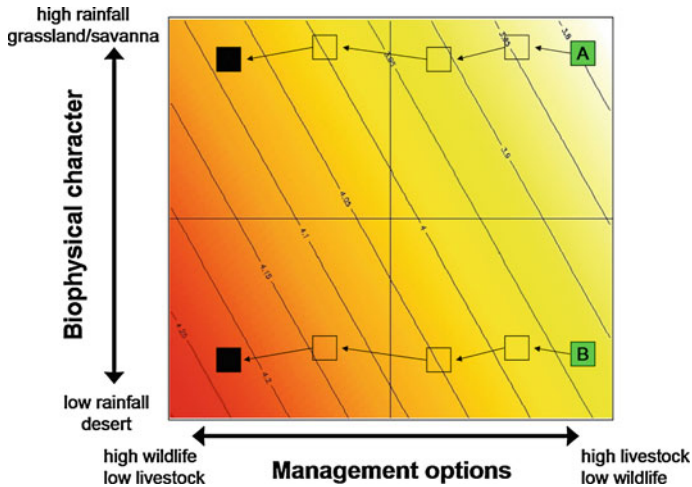
An important caveat to our results is that we examined financial benefits at the conservancy level. In reality, local households will vary in their ability to benefit from natural resources, due to variation in local power dynamics, skills and expertise, and personal histories (Botha et al. 2004; King 2007). An important avenue for future research, therefore, is to explore how the models developed here translate to the household or individual level. In addition, we also acknowledge that our data sets span a relatively small number of conservancies in

one geographic area of one country. It would be premature to generalize these results beyond our particular study area, and ideally our analysis would have had the luxury of a larger sample size that would have enabled more explanatory variables to be included in our model of financial benefits. We intend to address similar issues for all 52 conservancies in a subsequent manuscript that will allow larger models of conservancy income to be developed.

We have examined how wildlife composition affects financial returns of conservation to communities, but have not addressed here the larger question of how returns to wildlife conservation compare to land-uses such as agriculture or ranching. A rational profit-maximizing landowner will select the activity (or mix of activities) that provides the greatest net returns to their land. We would therefore expect that the legislation that allowed private landowners, and more recently, customary landholders, to derive benefits from wildlife would have led to a shift from agriculture to wildlife if this latter was indeed a more profitable form of land-use. Indirectly, evidence from the transect surveys we used here has shown that wildlife populations of a number of key species have increased substantially in the 2000s as compared to counts in the 1980 and 1990s (NACSO 2008), which we would expect under a hypothesis of increasing profitability of wildlife as a landuse. However, confounding factors such as environmental conditions could also explain this. We do not have comprehensive data on land under wildlife versus under ranching for Namibia and how this has changed over time. Personal accounts from ranchers, game reintroductions to both private and communal areas (NACSO 2008), and case studies that have compared financial benefits of wildlife versus farming in neighbouring countries (Barnes 2001) all suggest that an increasing proportion of land in Namibia is being turned over to wildlife. However, more research on the relative benefits of cattle ranching versus wildlife over a range of environmental and socioeconomic conditions is badly needed. It is also worth noting that, similar to other rural African communities, livestock in this part of Namibia is not only a productive asset that is sold on the market, but also a form of insurance and social standing. Strict cost-benefit analysis of livestock versus wildlife must be mindful that not all values associated with livestock can be monetized.

We have demonstrated that the structure of wildlife assemblages, as revealed through ordination methods, is a strong predictor of conservancy financial benefits in northwest Namibia. The implications for conservation are that managers may consider adopting more extensive, ecosystem-based management practices that facilitate the conditions necessary for diverse wildlife assemblages to persist on communal lands, rather than focus on intensive single species approaches to management. Not only is this style of wildlife management more beneficial for biodiversity as a whole, but it can result in greater financial returns to land.

From a conservancy standpoint, our results can be used by communities to assess their wildlife resources relative to other conservancies, and their progress, or not, towards a resource state that will generate greater levels of financial benefits (Fig. 5). It can also temper expectations for income gains over time. Conservancies can actively influence their large wildlife assemblages in the horizontal dimension by managing wisely, but can do little to influence wildlife assemblages along axis 2, as this axis is influenced by immutable factors such as elevation and precipitation. Hence, conservancies have the potential to “move” horizontally along ordination axis 1, but not vertically along axis 2 (Fig. 5). Consider Conservancy “A”, starting from a position in the upper right hand corner of Fig. 5, i.e., the area of lowest income generation. This conservancy has little wildlife on its land, which is mostly savanna. Managing for more diverse large wildlife assemblages will result in a gradual shift in position over to the left hand side of Fig. 5, i.e., towards higher income isoclines. However, no amount of management will be able to change its geographic qualities (biome, precipitation, etc), and therefore, conservancies that start in the upper right hand corner of Fig. 5 are unlikely to



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