Large mammal use of protected and community-managed lands in a biodiversity hotspot

N. Velho[1], U. Srinivasan[2,3], P. Singh[4] & W. F. Laurance[1]

1 Centre for Tropical Environmental and Sustainability Science (TESS) and College of Marine and Environmental Sciences, James Cook University, Cairns, Qld, Australia
2 Program in Science, Technology and Environmental Policy, Woodrow Wilson School of Public and International Affairs, Princeton University, Princeton, NJ, USA
3 Tata Insitute of Fundamental Research, National Centre for Biological Sciences, Bangalore, India
4 Researchers for Wildlife Conservation, National Centre for Biological Sciences, Tata Institute of Fundamental Research, Bangalore, India

Abstract

In large parts of the biodiversity-rich tropics, various forest governance regimes often coexist, ranging from governmental administration to highly decentralized community management. Two common forms of such governance are protected areas, and community lands open to limited resource extraction. We studied wildlife occurrences in the north-east Indian state of Arunachal Pradesh, where the Eaglenest Wildlife Sanctuary (EWS) is situated adjacent to community lands governed by the Bugun and Sherdukpen tribes. We conducted transect-based mammal sign surveys and camera trapping for mammals (>0.5 kg), and interviewed members of the resident tribes to understand their hunting practices and causes of wildlife declines. Interviews indicated hunting-mediated declines in the abundances of mammals such as the tiger *Panthera tigris*, gaur *Bos gaurus* and river otters *Lutrogale* and *Aonyx* species. Larger species such as *B. gaurus* were much more abundant within EWS than outside of it. Community-managed lands harbored smaller bodied species, including some of conservation importance such as the red panda *Ailurus fulgens*, clouded leopard *Neofelis nebulosa* and golden cat *Pardofelis temminckii*. Our findings show that protected areas may have important non-substitutive values but adjoining community-managed lands may also have important conservation values for a different set of species.

Introduction

Protected areas are often effective reservoirs of biodiversity (Watson *et al.*, 2014), decreasing species extinction risks (Karanth *et al.*, 2010), preventing damage to ecological communities (Laurance *et al.*, 2012) and maintaining essential ecosystem processes and services (Watson *et al.*, 2014). Despite this, only 12.2% of the global land area is under some form of protection (Chape *et al.*, 2005), and only 5.8% is strictly protected (Jenkins & Joppa, 2009). Large areas of natural habitat thus remain outside of formal governmental administration, and under alternative management regimes that may vary in their efficacy for biodiversity protection.

It is important to understand the role of various management regimes in conserving biodiversity, especially in the developing and tropical world, where most global biodiversity is concentrated (Schipper *et al.*, 2008) and species declines are occurring most rapidly (Dirzo *et al.*, 2014). One such approach is that of community-based forest management by resident groups or institutions, with varying degrees of governance and involvement (Bowler *et al.*, 2011). These are often typified by a local-scale participatory

administration that incorporates the rights of resident communities to extract natural resources (with a view towards poverty alleviation with greater participation of poor and marginalized communities; Agarwal & Gupta, 2005).

Despite the emerging importance of decentralization as a forest management approach in the developing world (Agarwal & Gupta, 2005), relatively little is known about the biodiversity value of such lands, especially in relation to (and in concert with) nearby protected areas. Assessments of the conservation efficacy of community-managed lands have yielded equivocal insights. For instance, community-managed forests and indigenous lands can reduce deforestation (Bray *et al.*, 2003; Nepstad *et al.*, 2006; Ellis & Porter-Bolland, 2008), increase forest biodiversity (Persha, Agarwal & Chattre, 2011) and may promote forest protection at large spatial scales (Mathur & Sinha, 2008). However, protected areas are generally more effective in stopping land clearing, but may fail to suppress more localized threats such as logging, hunting, fire and grazing (Bruner *et al.*, 2001).

A shortcoming of past evaluations of community-managed lands versus protected areas has been that comparisons are often geographically unmatched (but see

Nelson & Chomitz, 2011), making inferences about conservation efficacy complicated and difficult to interpret. Further, a traditional focus on differences in forest cover change or other anthropogenic disturbances between community lands and protected areas provides no direct information on the value of these management regimes for biodiversity conservation.

In this study, we assess the presence of mammal species in community-managed lands and an adjoining protected area within the same biophysical landscape in north-east India. We used transect-based animal sign surveys and camera trapping in conjunction with interviews to assess the persistence of a range of mammal species in both management regimes. We predicted that (1) because of human disturbances such as hunting, larger, potentially targeted species should respond more positively to protection than do smaller species; (2) for all species, abundance (a proxy that is reflected by the occurrence of animal signs) should increase with distance from roads; and (3) the perceptions of key informants should reflect observed patterns of species occurrence across the two management regimes.

Materials and methods

Study area

The study was conducted in Eaglenest Wildlife Sanctuary (EWS) and adjacent community-managed lands in the Indian state of Arunachal Pradesh (~20 km^2 each in EWS and community-managed land; Fig. 1). Arunachal Pradesh state borders Tibet to the north, Myanmar to the east and Bhutan to the west. EWS and its adjacent forests are part of the Eastern Himalaya Global Biodiversity Hotspot (Mittermeier *et al.*, 2004), and are the centerpiece of the
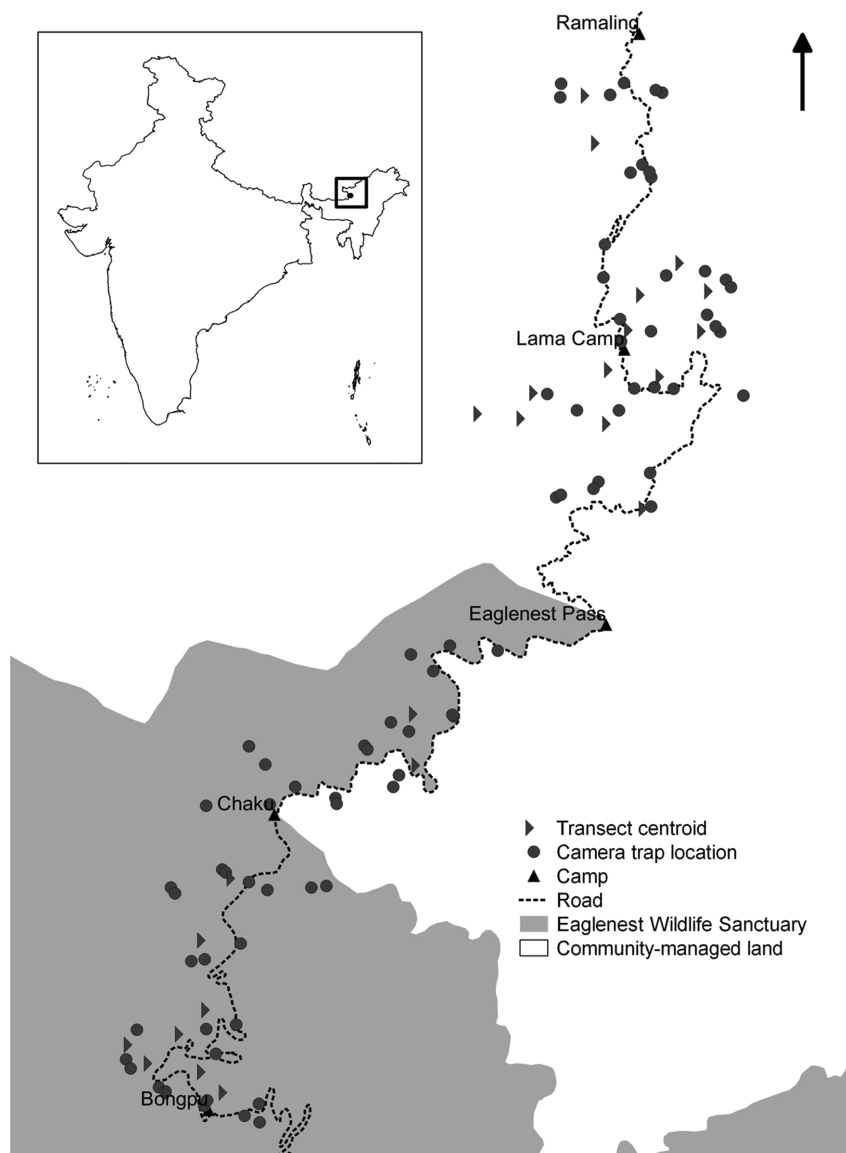

Figure 1 Map of the study area. The dark gray area is the Eaglenest Wildlife Sanctuary. The white area represents a part of the larger community land area that we sampled. Our sampling backbone was along the Foothill-Chaku-Tenga road (black line). Points represent camera trap locations, whereas the triangles represent transect segments for sign surveys.

3500 km^2 Kameng Protected Area Complex, the largest patch of contiguous forest in Arunachal Pradesh (Athreya, 2006). EWS has an altitudinal range of 100–3300 m asl and an average annual rainfall of 1500–3000 mm (Choudhury, 2003). Our study area varies in elevation from 1800 to 2800 m, with montane wet temperate broadleaf forest (Champion & Seth, 1968) as the dominant habitat.

Since 1989, EWS has been managed as a protected area by the Arunachal Pradesh Forest Department. Although under government administration, the Sherdukpen (or Shertukpen) tribal community consider EWS to be part of their traditional community lands. In the past, the Sherdukpen migrated annually through parts of EWS to barter essential commodities with other communities residing to the south in the plains of Assam (Fig. 1). With increased road connectivity and access to Sherdukpen villages and towns in the hills of Arunachal Pradesh, annual migrations to the plains are no longer essential for economic purposes and have almost entirely ceased.

EWS has had a history of anthropogenic use. Selective logging and road maintenance were carried out by the General Reserve Engineering Force (GREF) (Srinivasan, 2013), a road-building branch of the Indian Army. GREF staff and labor lived in camps and maintained the Foothill-Chaku-Tenga road, which was built in the 1950s (Fig. 1). This road, which is presently a single-laned, unpaved mountain road, is almost devoid of traffic and connects the higher altitudes and border areas of Arunachal Pradesh with the plains of Assam state. This road also remains impassable between June and September every year because of monsoon-induced landslides. The road is cleared for very low-intensity tourist traffic after every monsoon by the nature-based wildlife tourism management. In 1996, the Supreme Court of India passed an order that banned logging. This resulted in areas of regenerating selectively logged forest embedded within the primary forests of EWS.

Community-managed lands or unclassed forests are recorded as forests by the Arunachal Pradesh Forest Department, but are under the *de facto* control of resident tribal communities (Naniwadekar *et al.*, 2014). The community-managed lands adjacent to EWS are owned by the Forest Department (Chapter 4, Section 33 of the Assam Forest Regulation Act, 1891). Historically there has been no formal land-tenure system in the state, apart from the established hierarchy of ownership rights among tribes. The relatively recent Arunachal Pradesh (Land Settlement and Records) Act – 2000 tries to formalize land-tenure, but customary rights are still exercised by different tribes. Within their respective community lands, village councils exercise customary rights in forest areas demarcated by traditional boundaries.

We sampled community lands of the Singchung Village Council (Bugun tribe) and the Tukpen Village Council (Sherdukpen tribe). Land management practices within these community lands are variable. The Sherdukpen tribe has a larger populace with prominent clan-based ownership. Land managed by the Bugun tribe is typically owned by individual tribe members, and often leased to second- or third-generation Nepali immigrant families, cultivate tomato, cabbage, potato and kiwi, in return for part of the produce or an annual tax. Prior to the 1996 Supreme Court order banning logging, many Bugun and Sherdukpen people depended on timber harvests as an important source of income, resulting in areas of selectively logged forest. This heterogeneity in past and current land use has resulted in a mosaic of land uses in community lands, ranging from active cultivation, new and old fallows, older secondary or logged forest and primary forest (Athreya, 2006).

Culture and religion

Cultural and religious practices in the study area also vary, and might impact hunting practices and the forest management practices in the area. The Sherdukpen and Bugun practice Buddhism (Velho & Laurance, 2013), with the practice of Buddhism influenced by the visit of 14th Dalai Lama to these areas. Despite identifying themselves as Buddhist, certain animistic beliefs and practices co-exist and are embedded within Buddhist practices (Velho & Laurance, 2013). Although most Nepalis followed Hinduism (and a minority Buddhism) in the past, most have converted to Christianity over the last generation.

Sign surveys

To maximize how representative our sampling units were of the various habitats in our study area, and to ensure that sampling of community lands captured the spectrum of dominant land uses, we stratified our sampling at 14 sites across the three prevalent habitat types: agriculture and fallow ($\sim 1.5 \text{ km}^2$), secondary forests and primary forests (roughly equal in area). In EWS, which had only primary and secondary forest, we sampled 11 sites. Because our study area is extremely steep, with dissected terrain that limits physical accessibility, we used a segment of the Foothill-Chaku-Tenga road (Fig. 1) as the 'backbone' of our sampling effort. Within EWS and the adjacent community lands, a representative section of the road (averaging 10–12 km in length) was selected (Fig. 1). The road was then subdivided into 1-km long segments. In general, for community lands we had to sample distances further away from the road to find primary and secondary forests compared with EWS, although there was no significant difference [overlapping confidence intervals (CI)] between these two regimes in terms of average distance of transects from road.

At each 1-km segment, we established a 500-m long U-shaped transect with two 200-m long parallel sections joined by a 100-m long section. In some cases, we had to deviate around impassable topography, but not beyond a 20° angle from the transect bearing (unless confronted with dangerous topography or extremely steep slopes). Each transect was divided into 20-m segments. In each segment, we recorded any detectable mammal signs on or on either side of the transect (such as paw prints, hoofmarks, feces and scrapes) without deviating from the transect line. In effect, this meant that in the habitats we sampled, we were unable to record mammal signs at over ~5 m from the

transect line. Two observers walked along each segment 5 m apart, and recorded mammal signs independently. Information collected by both observers was used to estimate the number of transect segments occupied by each species by taking into account detection probability using the Lincoln–Peterson estimator (Seber, 1982). We also recorded any mammals that were identified visually or acoustically along each segment. Because we could not identify small carnivore signs to the species level, we treated small carnivores as a single guild. We also collected data on topography (slope), and habitat characteristics within 5×5 m plots positioned at 20-m intervals along the transect [using ordinal variables to estimate relative canopy cover (1–4) and understory cover (1–4), the number of trees >20-cm diameter-at-breast-height (DBH), DBH of the largest tree, the number of fallen logs, the presence of invasive plant species and any signs of human disturbance, such as cut stumps or vehicle tracks]. We conducted transect-based sign surveys in October–November 2011 and repeated them in August–September 2012. Because of a major landslide, we could not resample one transect in the community-managed land.

Camera trapping

Using the road as a sampling backbone, we conducted camera trapping in two adjacent blocks representing EWS and the community lands between April and June 2013. Each block was overlaid with a grid of 1×1 km cells. Within each cell, we deployed a Cuddeback Attack (Model 1149; Non Typical Inc., Green Bay, WI, USA; http://www.cuddeback.com) passive infrared camera trap in each of the 40 grids (22 in EWS and 18 in the community land). Average distance ($\pm$ standard deviation) between camera traps was 2.81 ± 1.48 km. Cameras were active 24 h a day, and the delay time was set to 1 s. Because we did not focus on estimating actual abundances under a mark-recapture framework, but were interested in the photo capture rate index from the camera-trapping exercise, the size of the grid cell (unlike the effort) is not foreseen to have any bearing on our results. We chose trap locations based on presence of tracks, trails and animal signs within each grid. We baited camera trap locations with a combination of rotting banana and smoked dried fish to maximize capture probabilities. Traps were attached to trees about 25 cm above ground to ensure that small animals were not missed. We moved each trap after 20 days to a new location within the same grid (therefore sampling two locations within each grid), and checked the traps every 5 days to ensure they were working properly. Our sampling effort was 856 trap nights in EWS and 677 trap nights in the community lands. Camera losses from theft, elephant damage and camera malfunctions did not allow us to precisely equalize sampling effort in the two management regimes.

Interviews

We conducted key informant interviews between July 2012 and August 2013. While choosing key informants, we used three basic criteria: their role in the community, their knowledge about wildlife and their willingness to speak to us. Village chiefs, youth leaders, teachers, local council members and administrative officers were considered as those having an important role in the community. Our approach was to first meet the village head and then other members who played an important role in the community. Once interviews were commenced, we asked interviewees to refer us to other potentially key informants such as hunters, former loggers and those involved in nature-based tourism.

Interviewees belonged to the Sherdukpen and Bugun tribes as well as second- or third-generation settlers from Nepal. We conducted interviews in Hindi, the most widely spoken language in the state. We were careful to ensure that no forest department personnel were involved while conducting interviews. Most people were willing to be interviewed (only five residents refused). In this way, we interviewed 99 residents from six large settlements near the periphery of EWS.

Our questionnaires were semi-structured, with open-ended questions to investigate hunting and wildlife taboos and restrictions. In addition, using photographs depicting the species occurring in EWS and the surrounding forest, we asked interviewees to assess population trends separately for each species over the last three decades. Responses were classified on a scale from minus two (for extirpations) to plus two (for large increases); a score of zero indicated no change. We also collected information on species status, threat perceptions, taboos, penalties and regulations related to hunting, illegal fishing and logging. Given that previous studies have shown that hunting plays an important role in daily activities of these people (Aiyadurai, Singh & Milner-Gulland, 2010), we also collected data on hunting motivations, patterns and methods, as well as preferred species. We also asked our informants if they believed that there was a higher abundance of animals in their community-managed lands or within EWS. Each interview lasted about 90 min.

Data analysis

We used Program R (R Development Core Team, 2014) for all analyses. We used logistic generalized linear mixed models (GLMMs) to investigate, for each species, the relationship between the detection-corrected proportion of transect segments with signs (henceforth, referred to as abundance) and our predictor variables. Across data from all transects and repeated sampling, we excluded five detections from the analyses: one each of Himalayan serow *Capricornis thar*, Himalayan black bear *Ursus thibetanus* and Asian elephant *Elephas maximus*, and two detections of Indian wild dog *Cuon alpinus*. Each of these were single records on a particular transect that were detected by only one of the two observers. We were therefore unable to calculate detection probability. Because repeated measures on the same sign survey were not independent of each other, we included the identity of the transect as a random effect in all models. Prior to creating a global model and candidate model set, we checked for associations among our predictor variables by

calculating correlation coefficients for every pair of the predictor variables and by examining variance inflation factors in the package *usdm* (Naimi, 2013) in Program R.

We selected *a priori* a global model in which abundance was a function of protection regime, sampling season, habitat disturbance, distance to road and species identity. Because we expected species-specific responses to differ, we also included interactions of species identity with protection regime, habitat disturbance and distance to road. We then created a candidate model set with all possible simpler subsets of the global model using the *dredge* function in the R package *MuMIn* (Barton, 2013). We selected models for inference based on Akaike's information criterion (AICc) corrected for sample size, which trades off model fit and model complexity (Burnham & Anderson, 2002). To explore the relative contribution of the fixed and random effects to the overall variance explained by the selected model, we used the *r.squared GLMM* function in the R package *MuMIn*, which calculates both a marginal R^2 (variance explained by fixed effects alone) and a conditional R^2 (variance explained by fixed and random effects combined; Nakagawa & Shielzeth, 2013). Finally, we used the *predictSE* function in the R package *MuMin* to generate predicted values from the selected model.

From our camera-trapping data, repeated captures of the same species at the same camera within a 1-h period were collapsed into a single record. These data were used to estimate species richness using the Jackknife 1 species-richness estimator, and community similarity using the Sorensen's, Morisita-Horn and Bray-Curtis dissimilarity indices. Sorenson's index calculates community dissimilarity based on species occurrence alone, whereas the Morisita-Horn and Bray-Curtis indices take into account the relative abundances of species as well. The effort-standardized number of independent captures of each species was computed and plotted as a function of body size.

From our interview data, most interviewees were unable to distinguish similar-looking species such as two macaques *Macaca* spp., smaller tree squirrels *Callosciurus* spp. and *Dremomys lokhriah*, and two species of river otters *Lutrogale* and *Aonyx* spp. The species within each of these groups were therefore pooled, and the average body mass for each group was used. We only included species for which there were more than 50 responses (excluding interviewees who did not know about these species or felt they did not occur in the forests they were familiar with). This approach yielded enough information to infer population trends over a three-decade period for 20 species. We then modeled the average scores in species increase, decline or no change (across all interviewees) as a function of species body mass (obtained from Menon, 2009).

Results

Sign surveys

Topographically, the protected area and community-managed land were similar. The median elevation in EWS (protected area) was 2149 m (95% CI: 1931–2574 m) and 2377 m (2115–2798 m) in the community land. Median slope was 22.8° (19.9–27.9°) in EWS and 24.4° (18.2–29.2°) in the community land. Median aspect in EWS was 215° (138–276°) and 226° (147–289°) in the community land. Transect-segment occupancy across all species was not spatially autocorrelated (Moran's *I* observed = 0.27).

Our predictor variables were not collinear (variance inflation factor <1.30, with correlations ranging from −0.12 to 0.17). Distance to the nearest village and protection regime (protected area and community land) was correlated because EWS has no settlements, but the community land does. Of these two variables, we retained protection regime as a predictor, but note that distance to village might also have a bearing on species occurrences and distributions. Our global model fit the data well (Pearson's *R* between observed and model-fitted values = 0.73). Of the 70 models in our candidate model set, the global model performed best (ΔAICc values for all models was greater than 7; Supporting Information Appendix S1)). The proportion of variance explained by our fixed effects (marginal R^2) was 0.43, and the cumulative variance explained (conditional R^2) along with the random effect (transect identity) was 0.46. From the second sampling of each transect, the response of each species was qualitatively very similar to that in the first season (Supporting Information Appendix S2 and S3).

Contrary to our expectations, most species did not show appreciable differences in abundance either with protection regime or habitat disturbance (Fig. 2). These include barking deer *Muntiacus muntjak*, Himalayan black bear and small carnivores that include yellow-throated marten *Martes flavigula*, golden cat *Pardofelis temminckii*, marbled cat *Pardofelis marmorata*, leopard cat *Prionailurus bengalensis* and Himalayan crestless porcupine *Hystrix brachyura*. However, as expected, larger bodied species such as the Asian elephant and Himalayan serow were more abundant in EWS than in the community-managed land, and also appeared to prefer more degraded habitats [$\beta \pm$ standard error (SE); protection regime: Asian elephant = −1.25 ± 0.32; Himalayan serow = −1.80 ± 0.64; disturbance: Asian elephant = 2.27 ± 0.36; Himalayan serow = 0.98 ± 0.52; Fig. 2]. Gaur *Bos gaurus* was most strongly influenced by protection and was much more abundant within EWS than in the community-managed land (protection regime: −4.39 ± 0.70; disturbance: 0.75 ± 0.36; Fig. 2). In contrast, wild pig *Sus scrofa* appeared to prefer more intact forests in community lands (protection regime: 1.75 ± 0.71; disturbance: −1.08 ± 0.90; Fig. 2).

Most species did not show appreciable differences in abundance with increasing distance from the road (for Asian elephant, Himalayan black bear, Indian wild dog, wild pig and barking deer, the range of $\beta \pm$ SE was −0.10 to 0.70, 0.70 ± 0.21 to 1.29). However, gaur (1.73 ± 0.32), Himalayan serow (2.18 ± 0.66) and small carnivores (2.07 ± 0.76) showed an increase in abundance with increasing distance from the road. Himalayan crestless porcupine was the only species that showed a higher occurrence closer to roads (2.32 ± 1.063; Fig. 3).
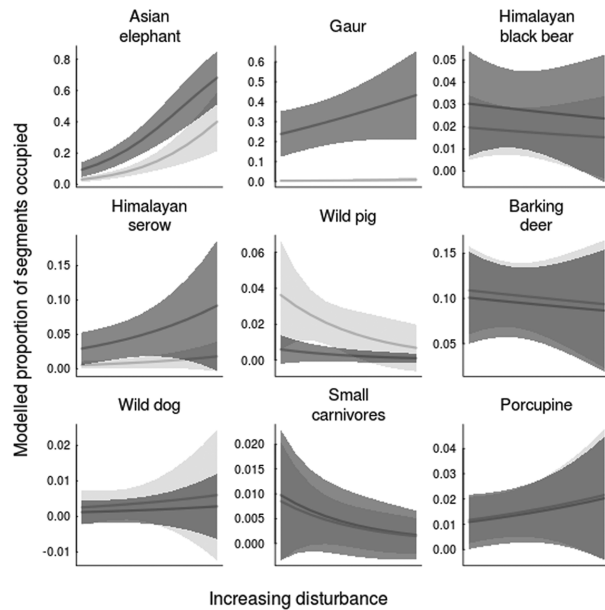
Figure 2 The relationship between proportion of segments occupied and habitat degradation in the protected area (dark gray) and community land (light gray). Solid lines represent fitted (predicted) values from the generalized linear mixed model, and the lighter polygons, the 95% confidence interval associated with the modeled predictions. From left to right and top to bottom, species are arranged in order of decreasing body mass.
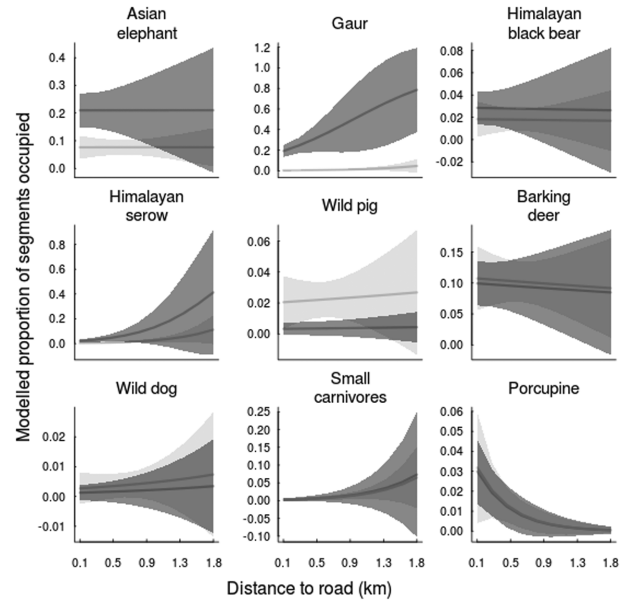

Figure 3 The relationship between proportion of segments occupied and increasing distance from road in the protected area (dark gray) and community land (light gray). Solid lines represent fitted (predicted) values from the generalized linear mixed model, and the lighter polygons, the 95% confidence interval associated with the modeled predictions. From left to right and top to bottom, species are arranged in order of decreasing body mass.

Camera trapping

Larger bodied species had more detections inside EWS compared with community land. For smaller bodied species, the reverse was found (Fig. 4). Species richness was comparable between EWS and the community lands (Jackknife 1 estimator: PA = 16.77 ± 2.54 SE, community land = 15.77 ± 1.88). However, these regimes did differ slightly in species identity (Sorenson's index = 0.25; with 0 = completely similar and 1 = completely dissimilar). Mammal communities across the two regimes were more dissimilar in terms of the relative abundances of different mammal species (Morista-Horn index = 0.43 and Bray-Curtis index = 0.60), indicating that while the same species may occur in both regimes, they had different relative abundances.

Interviews

A majority of respondents (91.9%) thought that more wildlife occurred within EWS than in community lands (5.1% were unaware of any differences and 3.0% felt that there was no difference). A majority (86.9%) also opined that wildlife was in a general state of decline (10.1% did not know and 3.0% thought there was no decline). Hunting was suggested to be the main reason for species declines (71.1%), followed by logging (38.4%), human population increase (29.3%) and militancy (14.1%). Hunting was predominantly carried out using guns (85.9%) whereas a few hunters used both guns
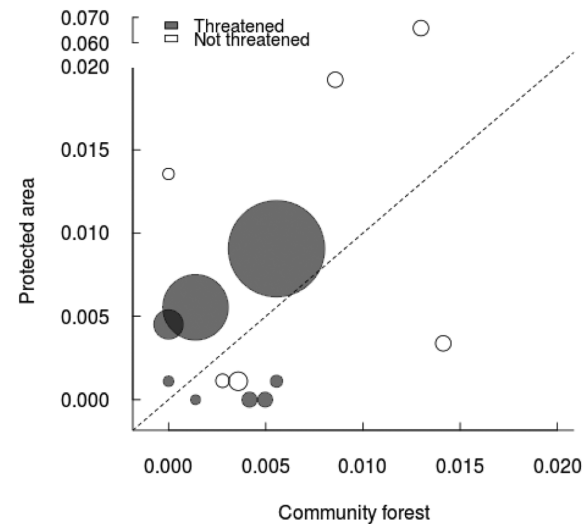

Figure 4 Relative abundance (effort-standardized number of photographic captures) of various species in community land versus protected area. The size of the bubble is proportionate to species body mass. The dotted line represents equal capture rates in community land and protected area. The bubbles above the line represent a greater number of photographic captures of species in Eaglenest Wildlife Sanctuary, and those below the line represent greater number of photographic captures in the community land.
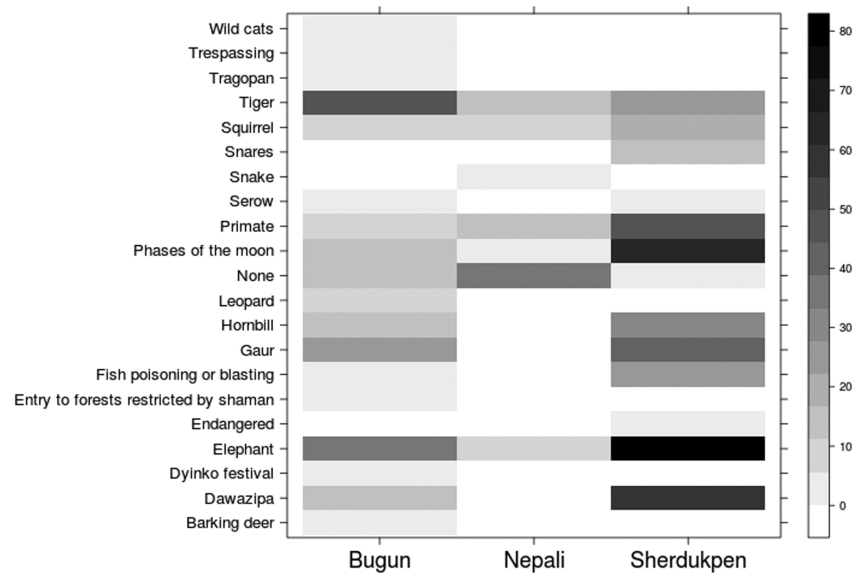
Figure 5 Percentage of respondents in three communities (Buguns, Sherdukpens and Nepalis) that reported a hunting restriction or a taboo.

and dogs (5.1%). Snares were still reported to be used (18.2%), despite formal restrictions against their use by some village councils. Wild pig, barking deer, Himalayan serow and goral *Naemorhedus* spp. were the most commonly hunted species. Hunters generally appeared to be selective; only 23% of our interviewees stated that they did not target any particular species and hunted all species.

Taboos and hunting restrictions were diverse across the three communities that live around EWS. Although species such as gaur are reported as a taboo, there was high variability across resident communities. These ranged from an absence of restrictions related to wildlife hunting to some taboos being relatively better known and followed (such as the restrictions on hunting tiger and Asian elephant; Fig. 5). Interestingly, interviewees did report drastic declines of tiger, which are a taboo species, as well as otters and leopard *Panthera pardus*, which were not reported as taboos. Asian elephant and *Macaca* spp. were perceived to have increased in abundance over time (Fig. 6). In general, interviewees reported greater declines in species abundances with increasing body size (except for Asian elephant, which are a taboo species; Figs 5 and 6).

Discussion

The role of community-managed forests in tropical conservation is recently gaining increased traction (Porter-Bolland *et al.*, 2011). Community-managed forests, unlike strictly protected areas, provide not only biodiversity benefits but also socio-economic benefits from both extractive use, and more democratic benefit sharing from activities such as tourism (Ellis & Porter-Bolland, 2008). Further, from a meta-analyses across the tropics, community forests had better governance, greater vegetation and lower and less variable rates of forest loss compared with protected areas (Hayes, 2006; Porter-Bolland *et al.*, 2011). Community-
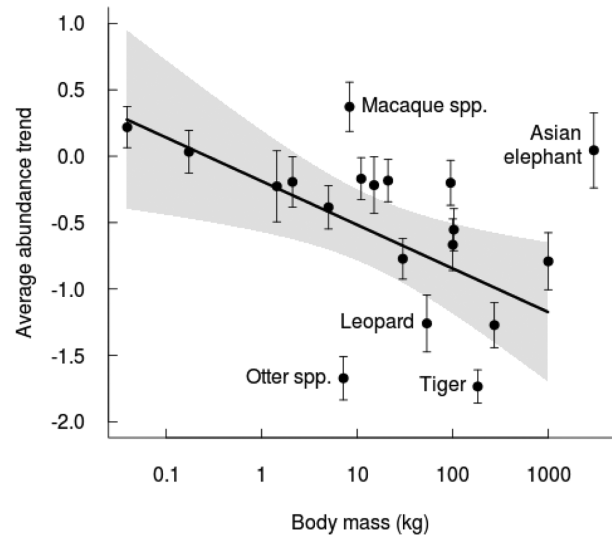

Figure 6 Perceived average species decline scores from key informant interviews. The solid line shows the fitted ordinary least-squared prediction for species with increasing body mass, and the gray polygon the 95% confidence interval of the prediction. Note that elephant is not part of the regression.

based and decentralized forest management may have an important role to play in maintaining tropical biodiversity along with the existing protected area network.

In our north-eastern Indian landscape, mammal species varied in their responses to forest management (protected area versus community lands), distance to road and habitat disturbances (Figs 2, 3). Although species richness did not differ significantly between EWS and the community lands, these regimes were dissimilar when a proxy for relative abundance (independent photographic captures) was taken into consideration. Body size appeared to be an important predictor of species responses, with larger bodied species

found more frequently within EWS whereas smaller bodied species were more abundant in community-managed lands (Fig. 4). Also, some species (e.g. elephant and serow) appeared to prefer more degraded forest, perhaps because of the abundance of bamboo (a food source) in EWS.

Based on our interviews of key community members, the overall perception of greater declines with increased body mass underscores the widely reported vulnerability of larger-bodied species (Fig. 6). Although there may be a taxonomic bias (large-bodied mammals are likely to be more detectable than smaller species), the accelerated declines of large-bodied species may be mediated by their intrinsic biological traits (low fecundity, long gestation period, etc.) in synergy with human impacts such as targeted hunting (Cardillo *et al.*, 2005). The case of gaur is especially noteworthy. Gaur is a large-bodied species (~1000 kg) that is nearly absent in community lands, and taboos against its hunting are rapidly eroding (Velho & Laurance, 2013); this highlights the importance of taking into account both species-specific traits and cultural contexts in conservation planning.

Perceived declines in abundances of several species are likely to be related to hunting pressure, which residents perceive to be a major threat to wildlife in the area as well as in other parts of Arunachal Pradesh state (Aiyadurai *et al.*, 2010; Velho, Karanth & Laurance, 2012). Interviewees reported dramatic declines of tiger, leopard and otters over the last three decades (Fig. 6). Although the killing of tigers remains a well-established local taboo compared with killing of leopards, this restriction may not to be effective enough to halt drastic declines of these species, which are targeted for poaching by non-residents (Velho & Laurance, 2013) for the illegal cross-border wildlife trade. Less obvious, but of equal concern, were the reported drastic declines of otter species, which have high illegal trade value. Although our data help identify a few species that are most vulnerable to illegal trade, more site-specific data are needed to support national and international wildlife trade databases (Toledo, Assmussen & Rodriguez, 2012).

Elephant and *Macaca* spp. were notable exceptions to the overall perception of declining wildlife populations. Restriction on elephant hunting is a strictly codified taboo in both community lands as well as EWS. Although our data on elephant occurrences do not show dramatic differences between EWS and the community lands, we do not have occurrence data on *Macaca* spp. to compare with the reported trends. However, interviewees reported crop raiding and agricultural losses to both elephants and *Macaca* spp. Whether these species benefit from human 'care takers' (through cultural mores) despite their perceived impact on crops needs further investigation (Lee & Priston, 2005).

One of the important findings we document here is the rich complexity of human interactions with wildlife, and how these interactions vary with ethnicity, space and time. For instance, in the last few decades, Nepalis have settled in community lands managed by the Bugun tribe, and practice intensive agriculture, changing the environments around EWS. This relatively recent immigration has also brought in much greater diversity in wildlife-related cultural practices

in and around EWS (Fig. 5). Studies from other parts of the Himalayas indicate that Tibetan Buddhism, which is also followed by the Bugun and Sherdukpen tribes, may play an important role in species conservation (Li *et al.*, 2014). Among the Ejagham tribe in Nigeria's Cross River National Park, cultural laws and codified beliefs and taboos have played an important role in the conservation of species such as the leopard; however, such taboos are in flux because of changing economies and departures from traditional belief systems (Jimoh *et al.*, 2012). Therefore, the cultural heterogeneity across various ethnic groups that reside around protected areas and how this varies with time needs to be considered when tailoring outreach programs that seek to reduce poaching pressures (Steinmetz *et al.*, 2014).

The biodiversity value of EWS and its surrounding areas cannot be overstated. This region may contain the second highest level of biodiversity in the world, after the northern Andes (Price, 2012). From our study, we argue that it is moot to debate whether EWS or its surrounding community lands have greater values for wildlife conservation. EWS harbors vulnerable large-bodied species and may provide an important refugium for hunted species. However, the community lands are important reservoirs for small-bodied species such as the vulnerable red panda *Ailurus fulgens*. In this context, the Bugun tribe is beginning to initiate a ~9000-ha community conservation area adjacent to EWS (Fig. 1). This would maximize the conservation values of the larger landscape, ensuring that EWS will not be isolated.

Although our study is from a single-paired site, an important perspective that arises from our study is that protected areas and community-managed lands can be complementary, protecting different species in different ways. In Arunachal Pradesh state, 62% of the forests are community managed and with growing populations and resource use, it is estimated that 50% of the state's forests will be lost by 2021 (Menon *et al.*, 2001). Similar trends are occurring elsewhere; for instance, human populations are growing rapidly in many of the world's 35 biodiversity hotspots (Bradshaw & Brook, 2014). In such contexts, protected areas are vital and non-substitutable, but community-managed lands could also play an important complementary role in promoting nature conservation, as is increasingly being recognized (Lemenager *et al.*, 2014).

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Supporting information

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Appendix S1. The candidate model set used in the GLMM analyses, with associated effect degrees of freedom, AICc and ΔAICc values. Columns 2–9 represent the predictors used in the model (with columns 7–9 representing interactions between predictors. A '+' sign indicates that the predictor (or interaction) was included in the corresponding model, and a '−' sign that it was not. Accordingly, Model 1, which includes all predictors and interactions, was our global model and performed best (ΔAICc = 0).

Appendix S2. The relationship between proportion of segments occupied and habitat degradation in the protected area (dark gray) and community land (light gray) at the second sampling session. Solid lines represent fitted (predicted) values from the GLMM, and the lighter polygons, the 95% confidence interval associated with the modeled predictions. From left to right and top to bottom, species are arranged in order of decreasing body mass.

Appendix S3. The relationship between proportion of segments occupied and increasing distance from road in the protected area (dark gray) and community land (light gray) at the second sampling session. Solid lines represent fitted (predicted) values from the GLMM, and the lighter polygons, the 95% confidence interval associated with the modeled predictions.