HUMAN BEHAVIOR AND BIODIVERSITY CONSERVATION IN TROPICAL SYSTEMS: MOBILITY, LIVELIHOODS, AND WILDLIFE CONFLICT AT MULTIPLE SCALES

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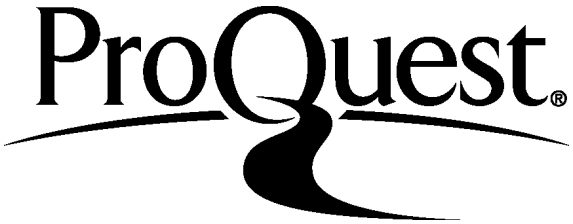

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ABSTRACT

Supporting human wellbeing and household livelihoods while conserving biodiversity and managing natural resources are challenges of global scale and importance. Across tropical regions, trade-offs between people and nature are most acute near the borders of strictly protected biodiversity areas. This dissertation consists of three studies that investigate the behavior or rural people, paying particular attention to mobility, how conservation strategies affect communities, and how people's livelihood decisions influence social and natural environments.

Chapter one examines a recent discussion debating the extent of human in-migration around protected areas in the tropics. One proposed argument is that rural migrants move to bordering areas to access conservation outreach benefits. A counter-proposal maintains that protected areas have largely negative effects on local populations and that outreach initiatives even if successful present insufficient benefits to drive in-migration. Using data from Tanzania, we examined merits of statistical tests and spatial methods used previously to evaluate migration near protected areas and applied hierarchical modeling with appropriate controls for demographic and geographic factors to advance the debate. Areas bordering national parks in Tanzania did not have elevated rates of in-migration. Low baseline population density and high vegetation productivity with low interannual variation rather than conservation outreach explained observed migration patterns. More generally we argue that to produce results of conservation policy significance, analyses must be conducted at appropriate scales, and we caution against use of demographic data without appropriate controls when drawing conclusions about migration dynamics.

Chapter two examines conservation strategies and the challenges of protecting biodiversity while also supporting household livelihoods. Across the tropics, efforts focus on balancing trade-offs in local communities near the borders of protected areas. Devolving rights and control over certain resources to communities is increasingly considered necessary, but decades of attempts have yielded limited success and few lessons on how such interventions could be successful in improving livelihoods. We investigated a key feature of household well-being, the experience of food insecurity, in villages across Tanzania's northern wildlife tourist circuit. Using a sample of 2,499 primarily livestock-keeping households we compared food insecurity in villages participating in the country's principal community-based conservation strategy with nearby control areas. We tested whether community-based projects could offset the central costs experienced by households near strictly protected areas (i.e. frequent human–wildlife conflict and restricted access to resources). We found substantial heterogeneity in outcomes associated with the presence of community-based conservation projects across multiple project sites. Although households in project villages experienced more frequent wildlife conflict and received few provisioned benefits, there is evidence that these households may have been buffered to some degree against negative effects of wildlife conflict. We interpret our results in light of qualitative institutional factors that may explain various project outcomes. Tanzania, like many areas of conservation importance, contains threatened biodiversity alongside areas of extreme poverty. Our analyses highlight the need to examine more precisely the complex and locally specific mechanisms by which interventions do or do not benefit wildlife and local communities.

Finally, chapter three examines how rural farmers and livestock keepers use mobility as an adaptive livelihood strategy. Continued migration to and within frontier areas is largely viewed

as a driver of environmental decline and biodiversity loss. Recent scholarship advances our understanding of migration decision-making in the context of changing climate and environments, and in doing so it highlights the variation in migration responses to largely economic and environmental factors. Building on these insights, this letter investigates past and future migration decisions in a frontier landscape of Tanzania, East Africa. Combining field observations and household data within a multilevel modelling framework, this letter analyses the explicit importance of social factors relative to economic and environmental factors in driving migration decisions. Results indeed suggest that local community ties and non-local social networks drive both immobility and anticipated migration, respectively, for different households. In addition, positive interactions with local protected natural resource areas promote longer-term residence. Findings are interpreted in light of how the migration literature understands changing frontier areas as they transition to human dominated landscapes. Doing so highlights critical links between migration behaviour and the conservation of biodiversity and management of natural resources, as well as how migrants evolve to become integrated into communities.

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CHAPTER 1: HUMAN MIGRATION, PROTECTED AREAS, AND CONSERVATION OUTREACH IN TANZANIA

1.1 Introduction

Human populations exist at greater densities and maintain higher rates of growth in areas of high biodiversity value than in all other inhabited areas (Cincotta & Engelman 2000; Williams 2011). A longstanding view holds that high human densities and persistent positive growth necessarily result in negative outcomes for tropical biodiversity both globally and locally (Meffe et al. 1993; Newmark et al. 1994). Accordingly, such anthropogenic threats are increasingly motivating biodiversity protection through the establishment of protected areas (PAs) and associated conservation outreach programs (IUCN and UNEP 2010; Roe 2008). Human in-migration to PA borders to access direct or indirect benefits provisioned by the PAs may increase existing anthropogenic pressures (de Sherbinin & Freudenberger 1998). However, though human migration represents a causal demographic process affecting population-environment dynamics in the tropics, it remains understudied (Bilsborrow 2002; de Sherbinin et al. 2008).

An emerging literature sheds light on in-migration to PAs (Wittemyer et al. 2008), yet questions persist concerning appropriate data sources and scales of investigation (Joppa et al. 2009) and the underlying mechanisms of migration processes (Scholte & de Groot 2010). In addition, the role of conservation outreach and development initiatives in PA-specific migration has not been adequately investigated, despite early concerns (Barrett & Arcese 1995; Noss 1997).

We examined rural population dynamics with respect to conservation outreach across the national parks network in Tanzania (Fig. 1). We sought to determine whether disproportionately high rates of in-migration to PA borders exist relative to overall levels of rural migration and

whether in-migration disproportionately affects areas receiving funding from the nation's conservation outreach program We also investigated the alternative explanation that (we prefer this wording as it explains our approach of testing an alternative hypothesis; if you disagree, then we accept you original edit) rural people move to access available and productive land and that explanatory evidence varies across such a diverse system. We conducted our analyses at a scale relevant to conservation policy in Tanzania. We used existing techniques from the literature investigating PA-migration dynamics—statistical tests and a spatially explicit method—and hierarchical modeling. Thus, we sought to move the discussion of PAs and migration forward to a policy-relevant context, as Wittemyer et al. (2008) recommend, and to redirect the focus of conservation science from simple measures of population to a more comprehensive analysis of demographic factors (Allendorf & Allendorf 2012).

1.1.1. Migration to Protected Areas

Wittemyer et al. (2008) found that high rates of rural population growth—their proxy of in-migration—exist at PA borders across Africa and Latin America, and these findings were challenged on the grounds that their data sources were incongruous and their methods inadequate (Shoo 2008). Joppa et al. (2009) used an alternative approach to examine in-migration around the same PAs and found no consistent pattern of higher growth rates. In addition, any conclusions regarding migration based on population measures such as growth rates alone without considering underlying fertility (thereby controlling for intrinsic growth) are problematic (Bilsborrow 2002). These issues of conflicting results and inferences from simple measures of growth bring into question the utility of large-scaled assessments in identifying whether in-migration exists near PAs. Regarding migration mechanisms, the most pointed criticism of Wittemyer et al. (2008) centered on the authors' interpretation of the factors driving high growth

rates: rural people migrated specifically to PA edges to access the benefits provided by the PA itself (Davis 2011; Igoe et al. 2008). This criticism reflected a decades-long debate over the impact of PAs on local populations and motivated a new framework for looking at how PAs affect migration behavior.

Scholte and De Groot (2010) summarized distinct mechanisms that could drive in-migration to areas near PAs as engulfment (people moving to access available land in low density areas where PAs are often established), attraction (people moving specifically to access PA benefits such as protected natural resources or employment from tourism), and incidental (PAs existing in areas of refugee resettlement or other displaced populations). In addition, demographers demonstrate that rural migration results from multiple factors at different scales acting on individuals and households (Barbieri et al. 2009; de Sherbinin et al. 2007), calling into question generalizations made at particularly large scales (Hoffman et al. 2011). For instance, results of site-based analyses of migration mechanisms near PAs largely refute that population growth is due to PAs simply attracting migrants. In Latin America and Africa, available and productive farming and grazing land, employment related and unrelated to PAs themselves, kin networks, and the development of adjacent urban areas drive a diversity of migration decisions for people settled near PA borders (Estes et al. 2012; Hoffman 2011; Zommers & MacDonald 2012). These mechanisms of migration represent the different models described by Scholte and de Groot (2010), each suggesting different approaches to alleviate further population increases and potential pressures on biodiversity.

1.1.2. Conservation Outreach, Migration, and Policy

Conservation outreach or integrated conservation and development programs, designed to mitigate the negative effects of PA establishment by provisioning benefit to local populations, were central to the attraction mechanism discussed by Wittemyer et al. (2008). The authors inferred that increased population growth rates resulted in part from in-migration of rural people to PA borders to access health centers, schools, infrastructure, and other benefits provided as part of conservation strategies. While outreach strategies designed to provide employment or market access may have distinct effects on migration from those provisioning health care or education, multiple approaches are typically combined (Brooks et al. 2006). Accordingly, for our analysis we define all such strategies together as outreach, recognizing that such initiatives can vary widely in the way they are integrated into conservation agendas (Borgerhoff Mulder & Coppolillo 2005).

In early studies of in-migration to PAs in Africa (Noss 1997; Oates 1995; Scholte 2003), researchers assumed outreach was successful in offsetting any costs incurred in living near the PA and that the net benefit to households would drive decisions to in-migrate. The conclusions of Wittemyer et al. (2008) relied on similar assumptions. Their findings are nevertheless puzzling—the suggestion of a global pattern of rural people moving to access benefits made available by PAs (e.g., outreach projects, employment, natural resources, infrastructure, security) is fundamentally contradicted by a large body of research detailing the significant costs of living near PAs (reviewed in West et al. 2006) even though it is clear that community services, such as hospitals and schools, may influence migration decisions in rural areas among some households (Barbieri et al. 2009; Lopez-Carr 2012; Massey et al. 2010). Moreover, the costs and benefits of living near a PA and reaping outreach benefits are not necessarily mutually exclusive, insofar as individuals, households, or communities may be differentially affected. Despite the widespread

existence of outreach programs, the long running debate on the overall effectiveness of outreach and related community-based conservation strategies—whether or not they actually provide social benefit or protect biodiversity and among which groups (Agrawal & Redford 2006; Naughton-Treves et al. 2005)—indicates the need for caution when considering their impacts on migration.

In sum, there are many factors that influence migration, puzzling contradictions between being attracted to the borders of PAs and suffering the costs of restricted resource use or human-wildlife conflict, and dangers associated with drawing inferences from incorrectly scaled analyses, especially when different explanations are not considered as alternatives. We therefore argue that more informed approaches to the specific relationship between outreach and migration are needed before making conclusions that have direct relevance to how and where conservation and community development practices are conducted (Hoffman et al. 2011).

1.2 Methods

1.2.1 Data Sources and Processing

We compiled demographic variables from ward administrative areas as their boundaries existed in 2002 (i.e., our unit of analysis was the ward; wards are clustered in districts). Human population growth rate, our outcome variable, was calculated from 1988 and 2002 census data from Tanzania's National Bureau of Statistics (NBS) and geographically referenced (see also Estes et al. 2012). These data were further processed in collaboration with NBS to correct errors from the original censuses, including identification of refugee camps and other features causing anomalous growth rate calculations. Wards in the Indian Ocean island districts of Zanzibar, Pemba, and Mafia were excluded because they did not include strict PAs gazetted during our

study period managed as part of mainland Tanzania's terrestrial PA network. The sample includes only rural wards containing populated areas. We also obtained coarse total fertility rates for 1988 from NBS. Finally, we estimated 1988 ward-level population density by resolving boundary changes with census documentation and maps from NBS and by accounting for unpopulated areas within PA boundaries. The resulting data set included 2,439 wards clustered in 118 districts.

To the demographic data we added geographically referenced boundaries for all 12 national parks established before 1988 (including Udzungwa Mountains, reclassified as national park from national forest reserve in 1992) (IUCN and UNEP 2010) that corresponded to the census period. Following previous studies, we classified wards adjacent to PAs as those within 10 km of the PA boundary, based on the distance feasibly traveled for daily livelihood activities (e.g., Bruner et al. 2001). To obtain a measure of conservation outreach effort, we collected data from Tanzania National Parks (TANAPA) financial records detailing individual project activities and funds. These data represented 366 individual projects carried out in 137 wards adjacent to PAs from 1994, when activities officially began, through 2002; funding was aggregated within wards. Tanzania National Parks most commonly funded schools and health centers, but projects also included roads, water infrastructure, and income improvement.

We modeled normalized difference vegetation index (NDVI) to produce measures of productivity across our period of interest from raw 8 km GIMMS NDVI measures (Tucker et al. 2005). The NDVI can be used as a proxy for ecosystem function and productivity (Pettorelli 2006), and the GIMMS data set has been employed over large spatial and longitudinal scales, including in East Africa (Duffy & Pettorelli 2012; Pelkey et al. 2003). We processed these longitudinal NDVI raster data (1988-2002) with the STARSPAN spatial processing tool to

estimate a time series of measures for each ward boundary unit (Rueda et al. 2005). Using a similar approach as Azzali and Menenti (2000), we then fitted periodic linear models to each time series to produce the most informative measures—mean, inter-, and sub-annual parameters—describing ward-level NDVI dynamics from 1988 to 2002. For further details see Supporting Information.

1.2.2. Analyses

We applied a hierarchical regression framework, hereafter referred to as the multivariate modeling approach. Using rural ward growth rate as our outcome variable, we defined an informed null model (M0) that included baseline population density, four NDVI covariates, and baseline fertility as predictors. Low population density and productive land describe geographic characteristics widely associated with rural migration, including in areas near PAs (Lopez-Carr 2012). In addition, including fertility rate so that in-migration was not confounded with intrinsic growth allowed for more informed inferences (Bilsborrow 2002). We evaluated hypotheses through interpretation of a series of increasingly complex models, along with their deviance information criterion (DIC) scores (model comparison strategy analogous to Akaike information criterion; see Spiegelhalter et al. 2002), through the successive addition of adjacency to PA (M1) and presence and funding of conservation outreach projects (M2) to the null model. The hierarchical structure of the models allows the level (intercept) and effects (slopes) of predictors to vary by district in order to account for unmeasured clustering effects; we refer to these as varying effects. Due to the non-Gaussian symmetric distribution of growth rate outcomes, we fitted models with the t distribution. Because we used this distribution in conjunction with a hierarchical structure, we applied a Bayesian approach (Gill 2008). In doing so, samples from the

joint posterior distributions of the parameters were generated by Markov chain Monte Carlo. Analysis was conducted in R (v2.15.2) with JAGS and the rjags package.

To further assess rural population growth rates near PAs, we applied a spatially explicit method similar to that of Joppa et al.'s (2009) used to re-evaluate nonparametric findings. We compared rural ward growth rates (converted to 1 km gridded values) at distances of 0-10, 10-20, 0-20, and 20-40 km around individual national park boundaries (hereafter, buffers). If in-migration to PAs exists, higher growth rates would be observed in 0-10 km buffers than in 10-20 km buffers and in 0-20 km than in 20-40 km buffers, assuming fertility and mortality are constant across adjacent buffer areas. Analysis was conducted in ArcGIS Desktop 10 and R. We refer to this method as buffer analysis.

We applied nonparametric statistical tests similar to those used in previous studies to determine if rural growth rates were disproportionately higher around PA boundaries or in wards with outreach projects than in other areas (Packer et al. 2011; Wittemyer et al. 2008). We did so with a Wilcoxon rank-sum test (wilcox.test function as part of the stats package in R).

1.3. Results

Across Tanzania, baseline population density and NDVI were the only predictors that had consistent relationships with growth rate outcomes. PA adjacency and conservation outreach initiatives did little to explain growth after controlling for baseline fertility rates.

Comparing results from a series of three multivariate models provided the most precise test of whether higher rates of in-migration occurred adjacent to PAs and, more specifically, whether higher rates were associated with conservation outreach activities. The multivariate model including controls and PA adjacency (M1) provided the best fit to the growth rate data; the

addition of outreach funding (M2 model) provided no improvement (Fig. 2). The M1 model indicated population density and NDVI measures were the only predictors that had consistent relationships with growth rate outcomes across Tanzania. While PA adjacency was included in the best-competing model, its ward-level relationships with growth rate varied substantially in both positive and negative directions so as to indicate no consistent effect when aggregated across all districts. In other words, none of the conservation measures was reliably associated with growth rate outcomes. We found a negative association between growth rate and density, based on the posterior mean and credibility interval for the effect of density in model M1. Of the four NDVI measures, growth rates were positively associated with mean NDVI from 1988 to 2002 and negatively associated with interannual variation. Because of the coarse measures of fertility, which existed at a similar level to that of the model's district-level varying effects, the model produced imprecise estimates for the effect of this predictor.

Varying effects estimates from the M1 model showed that unmeasured district-level factors contributed much of the variation in growth rates. First, varying intercepts estimates for the informative predictors indicated mean growth rate differed between districts (Fig. 3). Second, while pooled coefficients for baseline population density and mean and interannual NDVI were still significant when averaged across all wards (Fig. 2), the district-level slopes provided by the hierarchical model varied greatly (Fig. 3). In other words, the district-level adjustments made by the model to the coefficient estimates indicated significant variation between districts even though the coefficients themselves were significant across the country (Fig. 3). Therefore, the hierarchical structure of the model allowed the strength of each of the effects on growth rate to change depending on district, and in some cases the direction was reversed. The variation in these district-level effects was most pronounced for baseline population density (Fig. 3a).

Mean population growth rates in the 0-10 km buffers of all national parks were lower than rates within 10-20 km buffers (Fig. 4a, all PAs). This relationship held when 0-20 km buffers were compared with 20-40 km buffers (Fig. 4b, all PAs). In terms of individual parks, 9 of 12 had lower mean growth in 0-10 km buffers than in 10-20 km buffers. When comparisons were made for individual parks with 20 km-wide buffer zones, some parks showed different relationships with wider than with narrower zones (i.e., parks with lower growth rates in their 0-10 km than in 10-20 km buffers but higher growth rates in 0-20 km than in 20-40 km and visa versa). Despite these discrepancies, when comparing mean growth rate values across all PAs for both narrow and wider buffer zones the majority of individual parks had lower rates at their immediate boundaries.

Across wards, nonparametric tests did not support that growth rates are higher in wards within 10 km of PA borders than in non-adjacent wards ($W=149342$, $p=0.2456$). When considering conservation outreach activities, growth rates were slightly higher in wards where projects were conducted than in all wards combined ($W=90370$, $p=0.09258$), but the relationship did not exist when comparing project wards with those wards also adjacent to PAs but without projects ($W=5408$, $p=0.1548$).

1.4. Discussion

We found no convincing evidence of disproportionately high population growth rates due to in-migration to protected areas. The multivariate models indicated that low population density and land productivity were the two factors associated with in-migration, but they failed to show a positive relationship between migration and PA adjacency. Fertility must be considered when making these claims. Fertility is generally higher in remote rural areas, among resident and in-migrant populations alike (Carr et al. 2006), and PAs tend to be located in similar isolated areas

(Joppa & Pfaff 2009). Therefore, disproportionately high growth driven by fertility may confound conclusions about in-migration based only on growth rates. When fertility was included (despite considerable uncertainty about the size of its effect), we were able to discern an alternative explanation to PA-driven in-migration: rural people moved to PAs access available and productive land. In Tanzania, it has been documented that rural farmers and livestock keepers are primarily motivated by land resources when making migration decisions (e.g., Brockington 2001; Charnley 1997). More specifically, Estes et al. (2012) demonstrated that in-migration around Serengeti National Park was driven by access to available and productive farmland, which occurred as part of a larger migration process affected by resource constraints in origin areas as well.

The hierarchical structure of these models controlled for unobserved geographic effects across rural districts and showed there was variability in ward-level rates of in-migration and major factors influencing migration, despite populations generally moving to areas of fewer inhabitants and higher, less variable productivity. For example, differences in district-level varying effects (varying effects are a term specific to the methods/models used, AKA mixed or random effects; we included a short clause in Methods clarifying this point and are willing to elaborate) indicated that for some rural areas growth rate and population density were positively associated (Fig. 3a), which was inconsistent with the overall trend but suggests that in a small subset of districts people moved to more densely populated rural areas. This is not surprising considering that factors other than available land can influence rural-rural migration decisions. Similar but less variable relationships existed across districts between productivity measures and rates of in-migration (Fig. 3b, c). This need to account for geographic differences in Tanzania supports the idea that migration is a complex process and that people may move from and to rural areas for

diverse reasons (Barbieri et al. 2009; Zommers & MacDonald 2012). Therefore there are potential dangers associated with drawing general conclusions or national policy recommendations from locally based results because interdistrict variability is evident, just as problems arise from applying local recommendations drawn solely from large-scale findings.

Results from the analysis of the spatial relationship between PAs and growth rates supported the multivariate model outcomes. The majority of PAs had lower growth rates immediately at their borders than in more distant buffer areas, yet this relationship was not universal. Not surprisingly, geographic variation in social and ecological context across Tanzania suggested that different mechanisms potentially drive growth rates in different areas. In other words, even if higher growth rates were observed at PA edges than in nearby surrounding areas, therefore indicating the potential for PA attraction, park-specific interpretations must be made with caution.

For example, relatively higher growth rates within the 10 km buffer directly adjacent to Arusha National Park (Fig. 4a) may have been driven by the concentration of safari operators and associated tourism economy. This was an indirect form of PA attraction that Hoffman (2011) noted in Costa Rica. However, the relationship was reversed in the 20 km buffers around the same park (Fig. 2b). This result may be related to diverse patterns of land use, including grazing, smallholder farming, and plantation agriculture, in different areas around the park at greater distances from its borders. Such discrepancies may be more likely where buffers of increasing distance include more heterogeneous landscapes, which was potentially the case in Rubondo Island National Park, where the PA boundary was the island itself and where the nearby fishing- and trade-driven lakeshore development differed from inland agricultural areas (Bootsma & Hecky 1993; Nunan 2010). In addition, consistent patterns across the increasing buffer distances

may still fail to uncover multiple dynamics present around the same PA. Serengeti National Park appeared to experience lower growth rates in both its immediate 10 and 20 km buffer zones, yet closer analysis by Estes et al. (2012) show strong local variability in growth, reflecting different patterns of in-migration and land use conversion on the eastern and western borders. Lower rates may also be affected by PAs under different designation in the area, but we examined only populated rural wards subject to in-migration. Despite limitations, the buffer analysis corroborates (the edit appeared to change the meaning by making it unclear that both the buffer and multivariate analyses were from our results) results from the multivariate models that suggest no apparent PA-driven in-migration and demonstrate variation in the national parks network, which supports that multiple factors influence migration outcomes.

Results of our non-parametric tests supported the general finding of no in-migration to rural wards adjacent to PAs, but there are limitations to such tests applied to migration dynamics in general. For example, testing for a difference in growth rates based on a single classification (i.e., adjacency to PA) assumes the classification is unconfounded with all other factors determining growth rate, such as fertility, geographic, and edaphic features that might affect productivity. Further problems arise when simple tests are used in cross-national comparisons because census classifications (e.g., rural-urban, enumeration units) differ between countries (Bilsborrow 2002). Thus, although results of nonparametric tests agreed with those of our multivariate modeling and spatial analyses, we caution generalizing conclusions based simply on growth rates.

We found no convincing evidence of in-migration specifically to access conservation outreach resources provided by TANAPA (Fig. 2). Similarly, results of nonparametric tests did not suggest that wards with outreach projects had higher growth rates than other wards near national parks.

These conservation outreach projects were first implemented in 1988 in Serengeti National Park, and the program was formalized and expanded to all parks in 1994. The potential of such projects to attract in-migrants in Tanzania in general was previously noted (Barrett & Arcese 1995). Tanzania National Parks outreach supports specifically the same activities which have been used to explain a mechanism of PA-attraction (e.g., de Sherbinin & Freudenberger 1998; Oglethorpe et al. 2007; Wittemyer et al. 2008); communities themselves propose projects including most commonly the construction of schools and health centers, but also water and road infrastructure and financial support groups. Thus, it could be inferred such projects improve conditions thereby acting as potential pull factors (Scholte & de Groot 2010), though our conclusions do not support this mechanism of attraction in the context of national park outreach in Tanzania.

It can be argued, however, that benefits from outreach projects supporting broader community development may influence rural migration. For example, projects can provide jobs associated with the implementation of activities (Noss 1997). Yet in Tanzania, the national park-specific programs employ a nominal number of local staff (A. Mbugi, pers. comm.). More generally, although education, health care, and infrastructure can reduce the probability of out-migration (Barbieri et al. 2009; Massey et al. 2010), and lack of these services can contribute to out-migration from rural communities (Lopez-Carr 2012), there is as yet no clear evidence that people in-migrate to rural communities principally to access these facilities. The activities supported by TANAPA are substantial—in some parts of the country outreach funded the single primary school or dispensary in villages with no other services—but the outreach program as it is conducted does not provide sufficient benefits to drive in-migration. Similar shortfalls in

provisioning benefits from conservation organizations to rural people are observed across southern Africa (Scholfield & Brockington 2009).

In considering the relationship of PAs and outreach together with migration, our results contribute to the debate on migration to PAs and the effect of PA outreach (Igoe et al. 2008; Wittemyer et al. 2008): rural people in Tanzania likely make their decisions based on geographic factors related to farming and herding livelihoods rather than proximity to PAs and associated conservation activities. However, these findings must be interpreted within the limitations of our analysis insofar as livelihood activities themselves (e.g., settlement, land clearing, and overgrazing) can result in cumulative effects on land productivity (though we see no ward-level trend in NDVI; Supporting Information) and potential suitability for subsequent migrants. Furthermore, while we found that PAs and outreach in Tanzania did not directly attract migrants, we cannot deduce from our findings the ultimate effects of PAs and conservation initiatives on local livelihoods. For example, it remains uncertain whether, on average, PAs impose heavy costs that associated benefits including outreach cannot offset or whether PAs and outreach together impact livelihoods positively but without attracting migrants. Only direct comparisons of migrants' circumstances before and after a migration event (Salerno unpublished data) can address such questions. Our principal conclusion is that for migrants these conservation factors, whether positive or negative, are outweighed by the need for productive land to support rural livelihoods.

For example, in the Katavi-Rukwa ecosystem, an area with which we are familiar, in-migration is driven primarily by the search for land suitable for agriculture (Borgerhoff Mulder et al. 2007). Thus, observed population growth at PA borders is likely due to PAs being located in remote areas of available and productive land for livestock keeping and farming, regardless of park-

based conservation outreach activities. Though human settlement near PAs may still result in land change and isolation regardless of the attraction mechanism (DeFries et al. 2005), a more advanced understanding of the diverse types of migrants, their decision processes, and their varying impacts on resources (Zommers & MacDonald 2012) is necessary for effective management. The capacity of conservation efforts to understand these systems and translate knowledge into informed project design increasingly determines social and ecological success (Brooks et al. 2012).

Significant to the study of migration and conservation science, our results show the limitations and benefits of investigating in-migration around PAs with existing methods. We argue that hierarchical multivariate models with fine-grained units of analysis and controls for fertility produce results best suited for making conclusions about the relationships among PAs, conservation outreach, and migration at larger scales. Future research should include investigation of the drivers of migration in origin areas, the relationships between types of conservation initiatives and in-migrants, and how strict, multi-use, and community managed PAs differentially affect migration behaviors.

The implications of our work for conservation practice suggest that Tanzania's conservation outreach programs do not pose threats to protected biodiversity by attracting in-migrants or contribute to anthropogenic pressures through population increase. In addition, though people do not move preferentially to areas adjacent to PAs, if parks and reserves are located in areas of available and productive land, in-migration is more likely to result. Therefore, the management of PAs must be conducted within the wider landscape of human-occupied lands, based on appropriately scaled evidence.

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1.7. Figures

Fig. 1.1. Mainland Tanzania's national parks network.
Boundaries are as they existed in 2002 (black, national parks white with gray outline, administrative wards ; IUCN and UNEP 2010, National Bureau of Statistics 2002). Island areas not pictured were excluded from analyses.

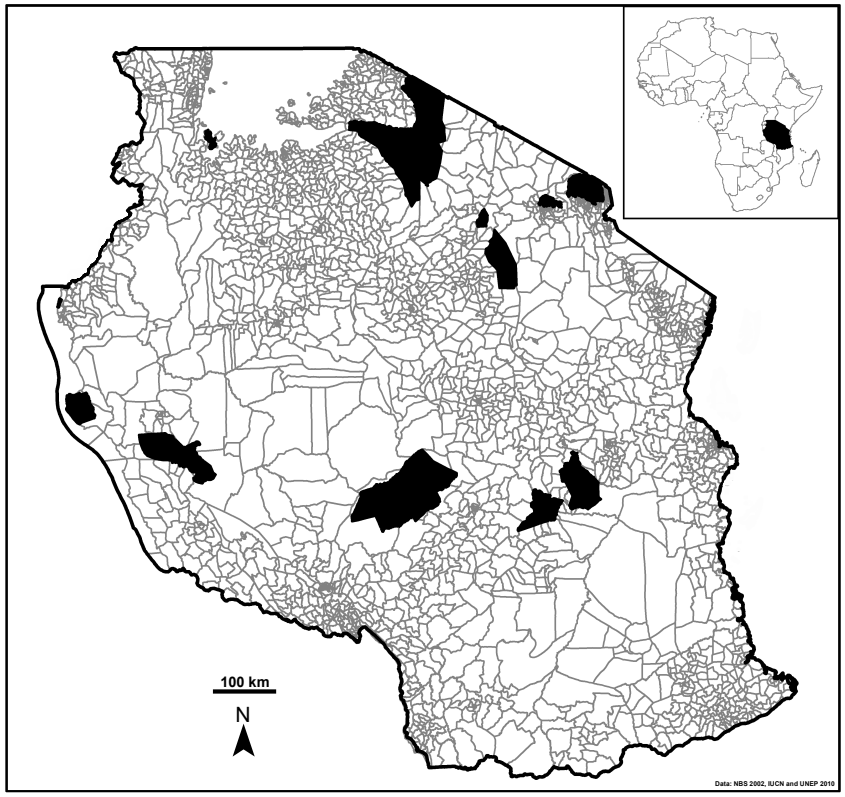


Fig. 1.2. Hierarchical model results for predictors of human population growth rate in rural areas.
Model results compared the null model (M0; population density, normalized difference vegetation index (NDVI) measures, and controls), M1 (including protected area adjacency and predictors of M0), and M2 (including conservation outreach projects and predictors of M1). Figure displays coefficient estimates of focal predictors excluding controls from the highest ranked model, M1 (deviance information criterion score (DIC) = -4556). M0 and M2 are not displayed; DIC = -4548 and -4547, respectively). Estimates for the geographic predictors (population density and NDVI measures) in models M0 and M2 are analogous to those of M1 (see Supporting Information). Posterior mean coefficients are plotted as points on the x-axis with 95% (gray bars) and 80% (black bars) credibility intervals.

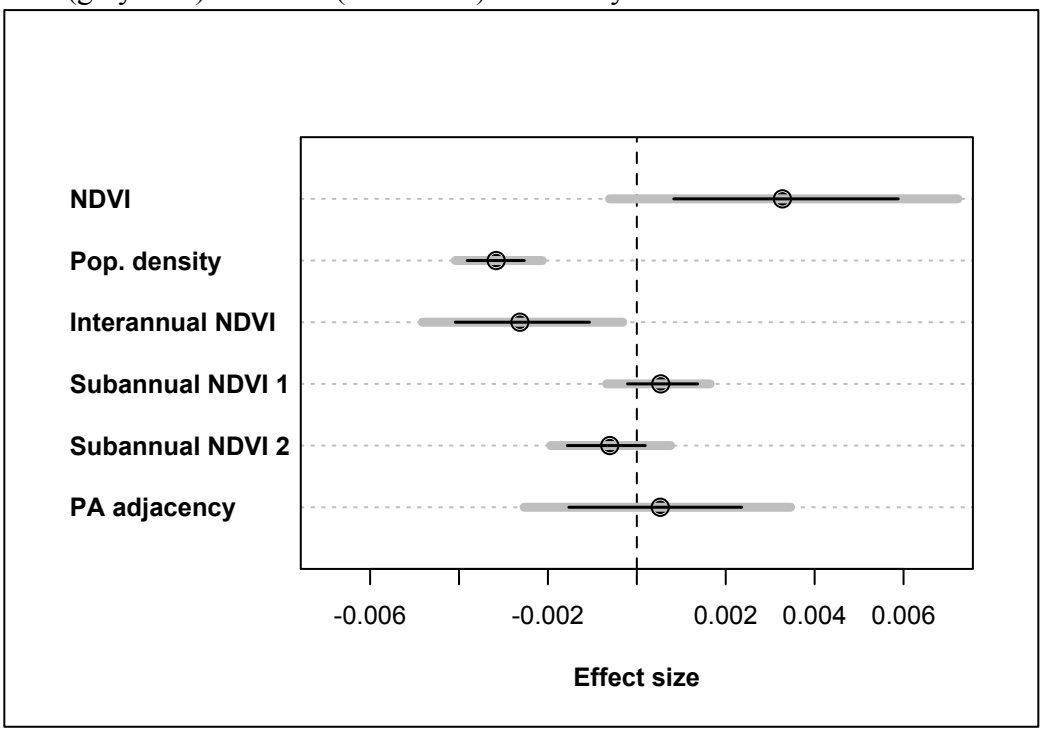


Fig. 1.3. Modeled district-level varying effects.
(a) Baseline population density (people/km2), (b) mean normalized difference vegetation index (NDVI), and (c) interannual variation in NDVI (point estimates of which are shown in Fig. 2) on human population growth from the best-fitting hierarchical model (M1) (black lines, mean effects pooled across the population; gray lines , slope and intercept estimates for each rural district).

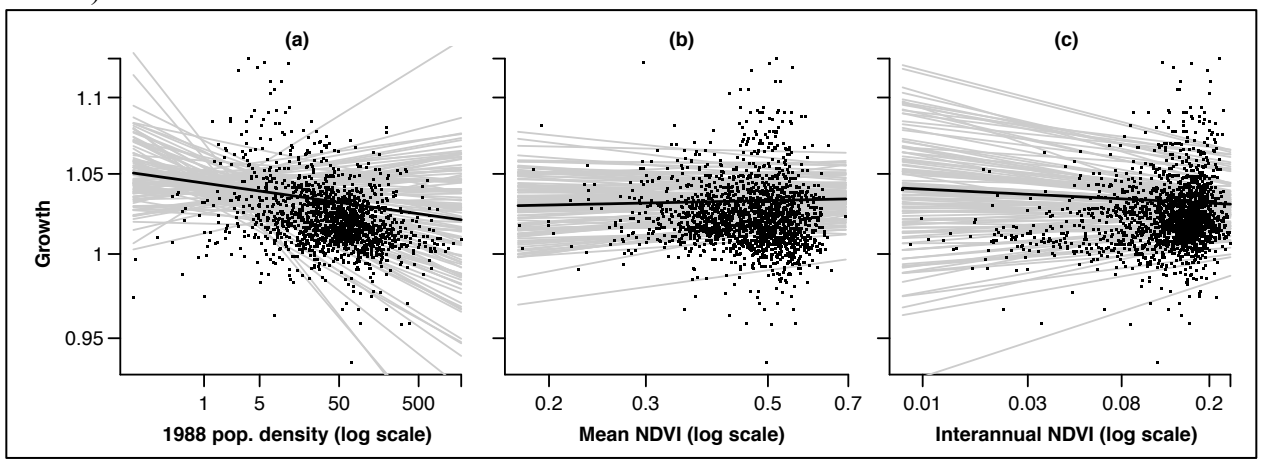
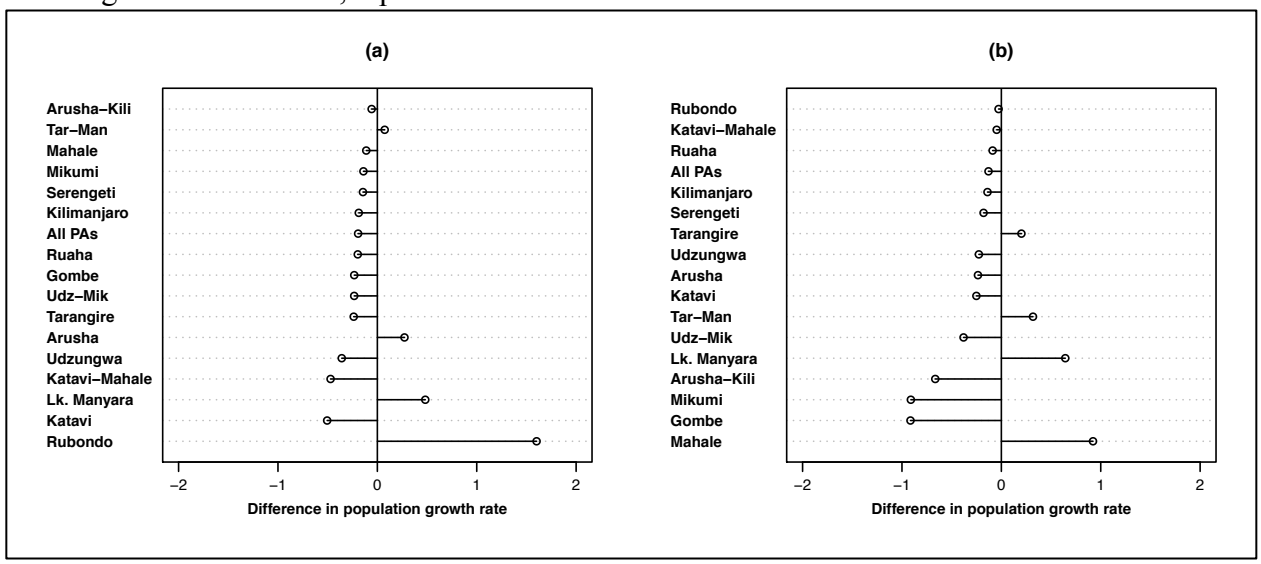


Fig. 1.4. Calculated population growth rate differentials at park borders.
Differences in (a) rural mean population growth rate between the area within 0-10 km and 10-20 km from protected area borders and (b) between 0-20 km and 20-40 km (positive values, higher growth immediately adjacent to the protected area (i.e., potential for attraction); negative values, lower growth immediately adjacent to the protected area). Comparisons are made for all protected areas together (All PAs), individual protected areas, and individual complexes where multiple protected areas existed within 80 km (Arusha-Kilimanjaro, Katavi-Mahale Mountains, Tarangire-Lake Manyara, and Udzungwa Mountains-Mikumi). Absolute values are sorted as the least to greatest difference, top to bottom.



1.8. Supplementary Information

Appendix S1. Description of normalized difference vegetation index (NDVI)

The literature on rural migration demonstrates that available and productive land primarily drive migration decisions. We use NDVI as a proxy for land productivity and suitability, which, in addition to population density and controls, allows us to model the potential for attraction of PAs and conservation activities in Tanzania. Here we discuss issues posed by the use of NDVI, how we addressed these issues, and the considerations that remain relative to our analyses and interpretation.

The AVHRR 8km GIMMS data (Tucker et al. 2005) we employ has been demonstrated as an effective measure of ecosystem productivity over large spatial and temporal scales in similar study systems; bimonthly composites provide information we expect to be relevant in the bi-modal annual climate patterns of Tanzania (e.g., Azzali & Menenti 2000; Duffy & Pettorelli 2012; Liu et al. 2013). These data were processed, including the removal of anomalous pixels and those indicating water or rock, using STARSPAN software (Rueda et al. 2005), and periodic models were then fit to each ward NDVI timeseries to extract the most informative components describing the productivity conditions (see Azzali & Menenti 2000). Each of these steps—processing and smoothing raw NDVI for error and effective use within analyses—are critical for producing accurate estimates of productivity (Pettorelli 2006). With these estimates included along with population density and fertility we created a null model (M0) of rural migration determined by available and productive land. To this null model of geographic and demographic determinants of rural migration we could then test our research questions regarding PAs and conservation outreach. Recent studies are effectively using NDVI within models predicting migration behavior (e.g., Hunter et al. 2013; Leyk et al. 2012).

Though we take steps to address the complexities of relying on NDVI for our analyses, potential limitations are worth noting. Specifically, we pay particular attention to the possibility of reverse causality within our modeling framework regarding NDVI and migration (see main text). That is, we use NDVI as a proxy for productive land which we hypothesize affects migration. However, we acknowledge that areas of productive land and early, rapid in-migration may in turn affect those areas' long-term productivity and, consequently, the measures of NDVI. Though changes in NDVI are principally determined by biophysical variables (e.g., temperature and rainfall) affecting vegetation, it is likely that increasing livelihood activities that clear or disrupt vegetation will also impact NDVI signatures over time. Recent research identifies land clearing for crops and grazing as likely influences on land degradation seen through declines in NDVI, though such relationships are less clear and weaker than temperature and rainfall effects (Liu et al. 2013; Thiam 2003). We account for some of this potential through our method used to produce multiple measures of NDVI over the study period. We allowed the periodic models to fit a ward-level trend over the timeseries (an overall increase or decrease in NDVI), but this term was not informative in describing the NDVI behavior and so was not retained in subsequent analyses. Though we have taken care to address the above issues, our conclusions must be read within appropriate context. Indeed, as with all applications of remotely sensed data, we attempt to apply the information in a way that is sensitive to the research system and specific questions at hand, and, in using NDVI as a proxy (as it is in our modeling framework), we are aware of the limitations when interpreting our results.

Appendix S2. Multivariate model results

We provide estimates of predictors of population growth rate from the second- and third-ranked hierarchical models. Posterior mean coefficients are plotted as points on the x-axis with 95%

(gray bars) and 80% (black bars) credibility intervals. Estimates of fertility are imprecise in both models and so not displayed.

Figure S2a. Null model (M0) including geographic predictors, DIC = -4548.

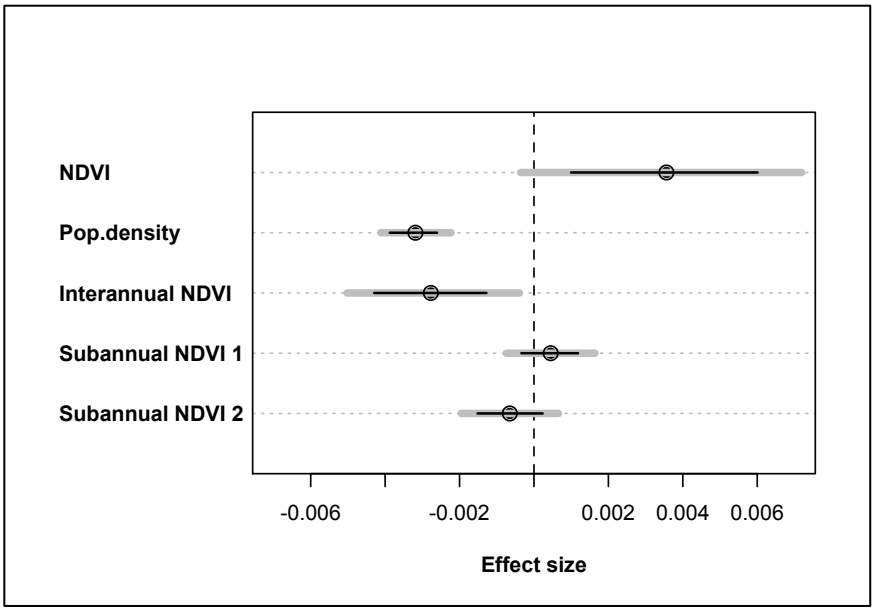
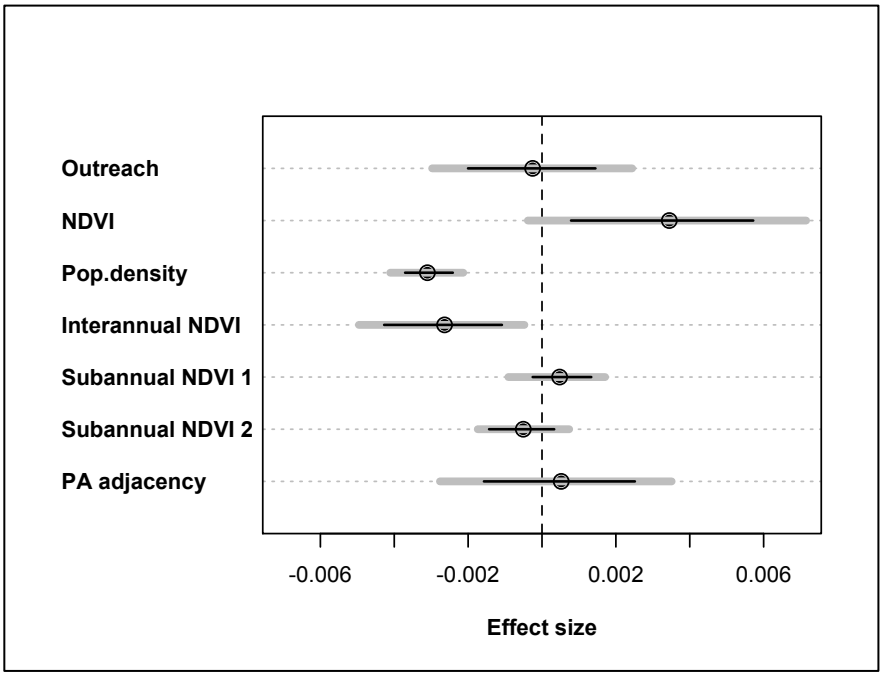


Figure S2b. Full model (M2) including geographic and conservation predictors, DIC = -4547.



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CHAPTER 2: HOUSEHOLD LIVELIHOODS AND CONFLICT WITH WILDLIFE IN COMMUNITY-BASED CONSERVATION AREAS ACROSS NORTHERN TANZANIA

2.1. Introduction

In tropical regions trade-offs between human well-being and biodiversity conservation occur most directly in rural communities near the borders of protected areas. Households in these communities incur costs through the loss of land and access to resources, which can limit livelihood opportunities (West et al., 2006; Coad et al., 2008), as well as from direct conflict with wildlife (Hulme & Murphree, 2001; Thirgood et al., 2005; Dickman et al., 2011). Community-based conservation, or the devolution of management and user resource rights to local communities, is promoted as a conservation tool to balance human well-being with biodiversity protection (Borgerhoff Mulder & Coppolillo, 2005). Since their inception the performance of community-based conservation and similar strategies has been debated, and policies in favour of such strategies have faced opposition from conservation biologists and social scientists alike (Naughton-Treves et al., 2005; Roe, 2008). Evaluations have highlighted numerous shortcomings: projects struggle to generate sufficient long-term revenue and maintain equitable access to economic benefits (Kiss, 2004; Garnett et al., 2007), to overcome assumptions regarding community institutions (Agrawal & Gibson, 1999) and state-level governance (Nelson, 2010), to ensure de facto control of resources at the village level along with adequate capacity support (Tallis et al., 2008), and to achieve ecological goals (Salafsky et al., 2001). Nevertheless, quantitative evidence suggests positive social and ecological outcomes can result from conservation efforts around protected areas (McNally et al., 2011; Ferraro & Hanauer, 2014b) and from community-based conservation projects (Tallis et al., 2008; Brooks et al., 2012).

Accordingly, it is important to consider comparative quantitative evidence in the evaluation of projects that engage in conservation efforts around protected areas (Pullin et al., 2013). Given the variation in outcomes demonstrated through evaluations of community-based conservation, one question remains central: are community-based strategies successful in achieving their broader goals? Decades of initiatives have yielded conflicting results, which can be attributed to multiple factors. The majority of evidence regarding project success is drawn from single cases over restricted time periods, rendering generalization problematic (Brooks et al., 2006). Desired project outcomes are rarely defined explicitly, which creates difficulty for evaluators in selecting appropriate measures of success (Agrawal & Redford, 2006), particularly when trade-offs are common (e.g. between ecological and economic goals; Brooks et al., 2012). Furthermore, insufficient attention is directed towards the intervening mechanisms by which projects and initiatives ultimately affect well-being and biodiversity (Ferraro & Hanauer, 2014b). Given the poor state of the evaluation literature in these respects (Miteva et al., 2012) (Botrill et al., 2014), and given that community-based conservation success necessarily depends on some level of community engagement affecting household livelihoods, we chose to investigate an outcome that directly affects well-being: household food insecurity.

Wildlife management areas, the dominant model of community-based conservation in Tanzania, were developed to improve livelihoods in communities living near strictly protected areas and in close proximity to wildlife. The model, common throughout southern Africa, was designed following the much-acclaimed CAMPFIRE and ADMADE programmes in Zimbabwe and Zambia (Leader-Williams et al., 1996). Similar national-scale policies are premised on decentralized control of resources, placing rights and authority over land and wildlife in the hands of local governments to incentivize effective local management and often to generate

tourism revenue (Igoe & Croucher, 2007). However, communities are rarely given the rights and capacity support to manage resources independently (Nelson, 2010). Instead, community-based conservation policy is essentially a financial bargain between national governments and communities, where a fraction of state tourism revenue is directed towards communities in exchange for local lands being designated for conservation.

In Tanzania and elsewhere, implicit in the bargain is that (1) potential for economic gain motivates local rule enforcement and conservation behaviour (e.g. poaching patrols, adhering to land use restrictions), (2) households receive direct or indirect benefits (e.g. cash, meat, access to schools, improved infrastructure), and (3) benefits outweigh the individual costs of human−wildlife conflict and loss of access to resources in the areas designated for wildlife (Mackenzie, 2012; Tetra Tech & Maliasili Initiatives, 2013).

The challenge in evaluating community-based conservation policies is therefore to test the performance of this bargain. Can economic instruments administered at the state level benefit household livelihoods and promote local wildlife management? In the absence of explicitly defined goals and outcomes (Brooks, 2006; Garnett et al., 2007) we evaluate the performance of wildlife management areas in terms of household food insecurity. This is a critical measure of well-being in Tanzania and in similar systems (see below). We pay particular attention to the role of wildlife management areas in potentially offsetting the costs of human−wildlife conflict; we propose that serving this function is important because of the necessity for households to tolerate interactions with wildlife for the wildlife management area programme to succeed. We conduct our evaluation across Tanzania's northern tourist circuit, using a household sample that includes multiple wildlife management areas alongside control areas. This approach addresses many of the problems of existing evaluations, and we target our findings towards the ongoing adaptive

management of community-based conservation in Tanzania and other strategies throughout the tropics.

2.1. Community conservation in Tanzania and the importance of food security

Across much of Tanzania and other dryland areas of southern Africa, rural livelihoods are based primarily on small farm and livestock holdings, with decision-making centred around avoiding chronic, seasonal and stochastic food shortages inherent in pastoral production and dryland agriculture (Ellis & Mdoe, 2003; Doss et al., 2008). Food insecurity in this region is widespread, stemming from numerous social and ecological factors, and the effects on children's health in particular have significant long-term consequences (Lawson et al., 2014). Flexible livelihood strategies, allowing households to manage mixed-asset holdings of livestock and/or crops, can be effective at reducing food insecurity (Hadley et al., 2007). However, restrictions on resource use and mobility imposed by the authorities of strictly protected areas in Tanzania's northern tourist circuit limit these strategies, and livelihoods are further strained by population pressure and environmental variability (Baird & Leslie, 2013; Goldman & Riosmena, 2013; Salerno et al., 2014).

In addition, conflicts with wildlife near these protected areas impose livelihood costs and contribute to household food insecurity. Attacks by predators on people (Packer et al., 2005) and livestock (Holmern et al., 2007; Røskaft et al., 2013), as well as crop depredation (Gillingham & Lee, 2003; Kaswamila et al., 2007), are frequent. Such conflicts inflict direct economic losses on livestock keepers and farmers, in addition to the opportunity costs of labour devoted to guarding herds and fields. Furthermore, there are direct links between livestock assets, livestock losses to

predators, and household resilience, demonstrated through short-term food insecurity and long-term child health outcomes (Hoddinott & Kinsey, 2001; Barua et al., 2013).

The performance of wildlife management areas in Tanzania has yet to be evaluated in terms of an underlying measure of household well-being, including the state of food insecurity. Research suggests that wildlife management areas experience problems common among community-based conservation initiatives, such as persistent human−wildlife conflict (e.g. Gillingham & Lee, 2003), difficulty distributing direct economic benefits within participating communities (e.g. Bamford et al., 2014), and barriers to representative local governance and gaining de facto control over resources (e.g. Benjaminsen et al., 2013). The majority of studies draw on single cases and varying, often qualitative indicators of success. Furthermore, a more comprehensive multisite report using quantitative measures demonstrated increased household assets over a 10-year period, along with signs of strengthened local governance (Tetra Tech & Maliasili Initiatives, 2013), yet no controls were implemented to account for underlying changes.

Outcomes of wildlife management areas are likely to be variable, but the variability must be quantified in meaningful ways to evaluate the policy as a whole. A systematic evaluation is needed, as wildlife management areas hold significant potential given the formal legislation of the rights to control the land (Sulle et al., 2011; Government of Tanzania, 2012), the significant economic value of wildlife for tourism (Sekar et al., 2014), the enabling environment of donor support, and the suggested gains in household wealth and village institutional capacity (Tetra Tech & Maliasili Initiatives, 2013).

We used data from an evaluation of rural villages both within and outside wildlife management areas in Tanzania to ask a critical question regarding the performance of the current community-

based conservation strategy: Is there an association between household food insecurity and wildlife management area status? Given the context of vulnerability across our system, we explored the association between food insecurity and human−wildlife conflict and asked whether wildlife management areas may lessen the negative effects of human−wildlife conflict, and whether certain livelihood activities can better support households that are vulnerable to food insecurity in the presence or absence of wildlife management area status. We assessed these research questions using data from 2,499 rural households and Bayesian multilevel statistical models.

2.2. Methods

2.2.1. Study area

The study was conducted within the protected area landscape at the centre of Tanzania's tourism economy (Sachedina & Nelson, 2010; Fig. 1). Significant areas of savannah and mixed woodlands are protected as national parks (prohibiting settlement, access and resource use by local people), game reserves (allowing licensed trophy hunting by non-local users) and the Ngorongoro Conservation Area (permitting limited settlement and pastoralist activities; IUCN & UNEP, 2010). Wildlife populations move within and between the protected areas according to variable seasonal rainfall patterns. Rural households of >60 ethnic groups (predominantly Maasai, Sukuma and Rangi) rely primarily on small farm and/or livestock holdings (Lawson et al., 2014), although pastoralists are increasingly shifting to more crop-based livelihoods (Baird & Leslie, 2013). Ecological and economic value along with ethnic diversity and widespread rural poverty motivate efforts by government and NGOs to balance trade-offs, thereby providing an ideal test case for evaluation of community-based conservation.

2.2.2. Household sample and data

Household data were collected between mid 2009 and mid 2011 in 56 rural villages as part of the Whole Village Project, a collaborative initiative of Savannas Forever Tanzania and the University of Minnesota (Borgerhoff Mulder et al., 2010; Lawson et al., 2014). The sampling of villages was based in part on the priorities of development agency partners and the permission of government officials, with attention given to representativeness across environmental and ethnic factors. Institutional assessments were conducted in each village through interviews with village leaders and randomly selected focus groups. In addition to qualitative information these activities yielded village-level variables describing the perceived effectiveness of village governments and relevant stakeholders (e.g. wildlife management areas, active NGOs; Supplementary Material 1).

We used a subset of 40 villages adjacent to protected areas to evaluate the effectiveness of wildlife management area interventions relative to a control group (Fig. 1). Adjacency was defined as within 45 km of a strictly protected area border, based on the furthest distance between a participating village within a wildlife management area and a protected area border. At the time of the study 13 of the 40 villages were participating in one of four wildlife management areas: Burunge, Enduimet, Makame and Makao (Supplementary Table S1). Participating villages contribute land to a jointly controlled wildlife management area, which is managed by an Authorized Association comprising representatives from each village. Although participation is voluntary at the village level, all resident households are considered to be part of the wildlife management area once the village joins the Authorized Association (Supplementary Material 1).

In each village a random sample of c. 60–70 households was selected from village administrative rosters. Tanzanian field teams administered structured questionnaires to household heads in Kiswahili or the local language. Given the salience of a reliable food supply to households in the arid and semi-arid zones of East Africa, we measured household well-being in terms of food insecurity, using the Household Food Insecurity Access Scale (Coates et al., 2007). Each household was assigned to one of four categories, food secure, mildly food insecure, moderately food insecure and severely food insecure, based on responses to a series of food-related questions. We treated food insecurity as an ordered categorical outcome variable. Human−wildlife conflict was recorded as a binary variable indicating whether or not a household had lost specific types of livestock and/or crops in the previous 12 months; crop loss frequency was not recorded in all villages, and therefore we report this variable for descriptive purposes only and omit it from our statistical models and main conclusions. Household productive livelihood assets were measured as the number of acres cultivated and number of livestock owned (cattle, sheep/goats and poultry). Models include a set of household control variables (e.g. construction material of home, bicycles and mobile phones owned) to account for differences in wealth as revealed through these commonly purchased items. The resulting database characterized the well-being and livelihoods of 2,499 households adjacent to protected areas (Supplementary Material 2).

The survey tools and methodology were approved by the University of Minnesota ethics board and collaborating Tanzanian research institutes.

2.2.3. Analytical approach

We considered descriptive measures of our household sample and then fitted multilevel regression models to investigate the relationship between food insecurity and the focal predictors used to evaluate our research questions: wildlife management area status, human−wildlife conflict, and livelihood strategy (i.e. variable reliance on livestock vs agricultural productive assets).

Because of the multilevel structure of the data (i.e. households clustered within villages, villages clustered within wildlife management areas) we included varying (random) intercepts for villages and wildlife management areas in all models. These intercepts capture contextual effects shared within villages and wildlife management areas that could make household-level food insecurity measures non-independent (Supplementary Material 3).

We fitted proportional-odds ordered logistic regression models using Markov-chain Monte Carlo methods (Plummer, 2003). In each model a linear equation gave the log-odds that a household was at or below food insecurity level c, as a function of predictors and control variables. We began with a basic model for μ containing only varying intercepts, along with a fixed effect for wildlife management area status:

$$\mu_{h,V,W} = \mathrm{A}_V + \mathrm{B}_W + \omega\chi_h$$

where A is the varying intercept effect for village V, B is the varying intercept effect for wildlife management area W, and ω is the fixed effect of wildlife management area status, with the status of household h indicated by the binary variable χ. The same equation μ applies at all levels c of the ordered outcome and is best understood as predicting more or less food insecurity (broadly

speaking) as the effects on the right-hand side change in sign and magnitude (McCullagh & Nelder, 1989, section 5.2).

To the basic model we added predictors derived from our research questions: wildlife management area participation, human–wildlife conflict, a wildlife management area-specific adjustment to the human−wildlife conflict effect, and productive livelihood assets. The adjustment predictors were specified as binary variables for three types of livestock loss along with interaction terms for each type of loss with wildlife management area status. The hypothesis that harmful effects of livestock loss are reduced by participation in a wildlife management area would be supported if model coefficients for livestock loss were positive (predicting greater food insecurity after a loss) but those for the interactions were negative (reversing the effect of loss, perhaps only partially). In addition to these predictors we added household-level control covariates to account for differences in wealth accumulation. All models, with the exception of the basic model, included household-level control covariates (Supplementary Material 2 & 3).

Questions regarding food insecurity, human−wildlife conflict and mitigating effects of wildlife management areas, and productive livelihood assets can be investigated by examining the signs and magnitudes of relevant predictors, as well as by comparing models containing various combinations of these predictors (e.g. models containing only main effects vs those containing interactions). We used a posterior predictive criterion, the log-conditional predictive ordinate (logCPOs; Gelfand, 1996), to compare and rank models (Supplementary Material 3). The most complex models we considered included varying predictor effects (random slopes) for individual wildlife management areas. However, logCPOs ranked these models relatively lower than simpler models. Unless otherwise noted, the results presented are based on estimates from the

top-ranked model, containing main effects and interactions for wildlife management area status and livestock loss, main effects for productive livelihood assets, and control covariates.

To describe the effects of focal predictors we present graphs showing how a change in each predictor changes the estimated odds of greater food insecurity. These changes are interpreted with respect to the indifference value of 1:1, indicating that a change in the predictor has little or no measurable effect. Uncertainty about the direction and magnitude of an effect is assessed from its posterior density (readily available via Markov-chain Monte Carlo samples). In the Discussion we interpret particular model results in light of our village-level institutional assessments to better understand contextual features that may inform management decisions for wildlife management areas.

2.3. Results

2.3.1. Food insecurity and participation in a wildlife management area

A higher proportion of households in villages that were participating in wildlife management areas were categorized as extremely food insecure, suffered from human−wildlife conflict (livestock and crop loss), and ranked lower on an index of household wealth, compared to control villages that were also adjacent to a protected area (Table 1; for comparisons between wildlife management areas see Supplementary Table S2). Although these results suggest that households in wildlife management areas fare worse than those in control villages, more careful analysis is required to evaluate the association between household well-being and wildlife management area status.

Our analyses did not reveal consistent differences in food insecurity between households in wildlife management areas and those in control villages (Fig. 2). On average, households in

wildlife management areas appear to be more food insecure (Fig. 2) but there is considerable variation between areas. We found more severe food insecurity in two of the four management areas (Enduimet and Makame; 43% of all households in a wildlife management area), whereas levels of food insecurity in the remaining two areas (Burunge and Makao) were similar to or lower than in control villages. The odds of severe food insecurity for a typical household in Makame were 5.67 times greater (95% posterior credibility interval: 2.15−12.36) than for a similar household in a control village, whereas the odds for households in Burunge and Makao (Fig. 2) were similar to those of households in a control village. Wildlife management area status per se is therefore not a reliable predictor of food insecurity, but households in certain wildlife management areas may tend to be more or less food insecure.

2.3.2. Wildlife conflict and participation in wildlife management areas

Nineteen percent of sample households experienced one or more predation events on medium or large livestock in the 12 months preceding interviews, and the extent of these losses varied with wildlife management area status (38 and 10% of households in villages in wildlife management areas and control villages, respectively; Table 1). This pattern was in households reporting predation on poultry. Our models supported an association between livestock loss and more severe food insecurity, accounting for differences in livestock holdings and other factors (Fig. 3, dashed lines). Losses of cattle and sheep/goats were both associated with greater food insecurity, although there is more uncertainty regarding the deleterious effect of cattle loss, indicated by its relatively broad posterior density.

Despite these costs, results indicate that wildlife management area status was associated with a reduction in the harmful effects of livestock loss. Figure 3 shows the main effects of livestock

losses (dashed lines) along with adjustments to those effects experienced by households in wildlife management areas (solid lines). Each plotted adjustment combines a main effect of loss with a wildlife management area interaction, leading to a modest reversal of the effect of loss. Losses of sheep and goats in households in control areas, for example, increased the odds of severe food insecurity by a factor of 1.69 on average (1.13−2.45), whereas similar losses in households in wildlife management areas increased the odds by 1.44 (0.94−2.11). The latter interval includes the indifference value 1:1, suggesting the possibility that losses of sheep and goats in households in wildlife management areas may have little or no relationship to food insecurity. The wildlife management area adjustments for cattle and poultry show the same pattern but are more uncertain, although the density for cattle loss displays a substantial shift (Fig. 3). We attempted to differentiate the effects of individual wildlife management areas by allowing interactions to vary, but these more complex models ranked lower using logCPOs (model m2, Supplementary Material 3), and the wildlife management area-specific effects did not reveal between-area differences. These analyses were unable to detect specific mechanisms through which wildlife management areas could mitigate for livestock loss (see Discussion).

2.3.3. Food insecurity and productive livelihood assets

As could be expected, households with fewer productive livelihood assets (i.e. livestock and cultivated land) tended to be more food insecure (Supplementary Fig. S1). This relationship was consistent for sheep and goats, poultry, and cultivated land, whereas fewer cattle were not associated as consistently with more severe food insecurity. A lower-ranked model including an interaction effect between productive livelihood assets and wildlife management area (model m3, Supplementary Material 3) suggested the effects of these assets did not differ between households in wildlife management areas and those in control areas. Thus, owning additional

assets (particularly sheep/goats, poultry and cultivated land) is equally effective in protecting households against food insecurity whether or not a household is located in a wildlife management area. As with human−wildlife conflict and interaction effects noted above, a more complex models that allowed the effects of productive assets to vary by individual wildlife management area (model m4, Supplementary Material 3) ranked lower and produced estimates with more uncertainty.

2.4. Discussion

We first discuss specific results in the context of the Tanzanian wildlife management area experiment along with the implications for national conservation policy, integrating qualitative institutional assessments from each village into our quantitative findings. We then discuss the broader significance of our study.

2.4.1. Are wildlife management areas working?

Testing the economic arrangement of wildlife management area policy demonstrates that the strategy is not a universally effective community-based conservation solution, and it is likely that multiple factors contribute to the diversity of observed outcomes. We show that two of the wildlife management areas studied may support household well-being, but households in the remaining two are particularly food insecure. This variation in effectiveness between wildlife management areas is most likely a result of institutional differences at the wildlife management area level, which we observed after controlling for other sources of variation between households (e.g. wealth) and between villages (e.g. social and ecological context, Supplementary Material 3; Goldman & Riosmena, 2013; Lawson et al., 2014). Multiple factors contribute to the ability of the Authorized Associations governing wildlife management areas to meet programmatic and

institutional challenges. These include protecting wildlife and habitat, attracting tourism companies and negotiating contracts (increasingly difficult for remote, dry and less biodiverse areas), collecting earned revenue from the state, distributing revenue to participating villages, and providing direct payments and adequate farming and grazing land to households. Furthermore, their capacity is affected by the varying levels of reliable support from donor organizations and agencies (see also Tetra Tech & Maliasili Initiatives, 2013). These challenges are compounded by frequent incidents of human−wildlife conflict.

Because of the significant costs wildlife can impose on households, we explored associations between food insecurity and human−wildlife conflict and found suggestive evidence that the costs of losing livestock may be slightly lower in wildlife management areas. The interaction effect, although weak and imprecise, is evident when averaged across all wildlife management areas in the sample, and is consistent for various types of livestock lost. Evidence for such a mitigating effect is important, particularly if it is widespread, because the success of wildlife management areas depends on the presence of large wildlife populations to attract tourism revenue, but more wildlife leads to greater losses of livestock and crops, and ensuing food insecurity (Kaswamila et al., 2007). Identifying where such mechanisms exist will be an essential step in understanding which features of the intervention are effective and which are not.

We can only speculate as to the underlying effects responsible for food security outcomes and the observed reduction in the costs of human−wildlife conflict in wildlife management areas. Implicit in the wildlife management area bargain is that state-level revenues will, through various assumed pathways, positively affect residents. Wildlife management areas provide tangible benefits, such as the construction of health centres, schools and wells, implemented through the Authorized Associations, and most participating households recognize these gains (Tetra Tech &

Maliasili Initiatives, 2013). Perhaps, as suggested by Baird (2014), development initiatives may be concentrated in and around protected area networks, such that external organizations and funders preferentially direct resources near wildlife management areas. Such forms of rural development affect household health but our data are inadequate to test a causal link between development and food security.

Less direct benefits of wildlife management areas establishment may emerge as a consequence of direct engagement with the policy process, and such institutional developments may distinguish wildlife management area villages from other villages that face similar challenges. The lengthy authorization requirements include the establishment of village land-use plans that designate areas for wildlife along with village forests and pastures (Sulle et al., 2011). Well-managed resource areas benefit households, and villages typically receive support from external organizations working with the wildlife management area Authorized Associations, although we do not have the data to address these claims directly. External donor support may strengthen the capacity of local governments (e.g. Persha & Andersson, 2014), and the aforementioned multisite report (Tetra Tech & Maliasili Initiatives, 2013) identifies small but tangible gains in the management capacity of wildlife management area village governments and shows that residents increasingly hold their leaders accountable. It is noteworthy, however, that the negotiation of rights to village lands and wildlife can also result in the appropriation of these rights by external interests or local elites (Benjaminsen et al., 2013). Also, there is potential for financial gain from increased tourism development (Ferraro & Hanauer, 2014b), although local job creation by wildlife management areas is rare.

It is unlikely that a single explanation accounts for how households benefit from wildlife management area participation. We highlight the variability in our observed outcomes, which

suggests that the wildlife management area bargain holds potential and that the mechanisms by which households can be affected may vary across contexts.

2.4.2. Insights from village leaders

Assessments conducted with community leaders and administrators may in part explain why Enduimet and Makame wildlife management areas appear to suffer particularly severe food insecurity, whereas Burunge and Makao are relatively secure. Leaders in the villages of Makao reported the highest earnings from conservation activities, followed by those of Burunge, whereas the two villages in Makame stated that they had yet to receive any revenue from the wildlife management area. Leaders in Makao described benefits from the wildlife management area, such as funding the construction of a secondary school, staffing a village dispensary and facilitating cooperation between the villages, although there was often a lack of consensus on whether revenue was generated by the wildlife management area itself or if funds came from other sources, such as NGOs or tourism companies. Leaders in one of the Makame villages agreed that empowering the community to manage and benefit from their natural resources was a gain, but observed that 'areas for people to graze their livestock are being reduced, [and management has] failed to supervise natural resources efficiently' (Ndedo village leader; interviewed 27 February 2010).

Village assessments revealed that the wildlife management areas, as institutions, varied in their ability to generate revenue and manage resources, following the pattern of observed differences in food insecurity between areas (Fig. 2). The precise reasons, if causality is indeed entailed, are uncertain. The four wildlife management areas vary across ecological gradients, affecting their potential for tourism, although we attempted to control for such variation in our analysis.

Differences in revenue may be implicated, although these were reported by the village leaders and may therefore reflect perceived rather than actual earnings (and may be subject to misuse). In terms of institutional gains, village leaders in each of the wildlife management areas cited improved management capacity and engagement within and between village governments as positive outcomes of participation, which was suggested in the multi-site report by Tetra Tech & Maliasili Initiatives (2013). Although these benefits of wildlife management areas are evident at the village level, links to household benefits remain unclear.

On the basis of our findings we recommend that Tanzania's national conservation policy take into account the variability in the performance of wildlife management areas. Some areas may be moderately successful despite disproportionately high levels of human−wildlife conflict, and revenue generation is a likely precursor for mediating any benefits to households. If the wildlife management area programme is to continue, we recommend that implementing agencies think carefully about the specific pathways by which revenues and benefits are transferred from wildlife management areas to participating villages and households, the aspects of livelihoods and well-being that are likely to be affected, how to monitor these outcomes and how they vary between wildlife management areas and villages.

2.4.3. Broader implications

Variable outcomes of wildlife management areas are not entirely inconsistent with the majority of site-based studies that have shown these areas to be largely ineffective (e.g. Kangalawe & Noe, 2012; Benjaminsen et al., 2013). It is possible, however, that case studies tend to be conducted in villages with known tensions (Borgerhoff Mulder & Coppolillo, 2005). Detailed accounts of how projects fail are useful, but generalizations regarding the overall effectiveness of

a national-scale strategy should be drawn with care. Our failure to identify consistent positive or negative outcomes of community-based conservation across multiple projects and a fairly broad landscape in Tanzania is consistent with reviews and empirical work globally, which indicate that outcomes of community-based conservation are variable and context-dependent (Garnett et al., 2007; Tallis et al., 2008; Brooks et al., 2012). Thus it is inadvisable to make recommendations regarding policy or implementation on the basis of single case studies. Analyses employing large sample sizes across sites, with multiple controls, are necessary to distinguish outcomes of specific interventions from confounding factors.

We have described the challenges of evaluating community-based conservation. We recommend the use of a meaningful measure of household well-being, sampled across project and control villages, to assess the strategy performance of community-based conservation in the absence of clearly defined project goals and outcomes. However, we acknowledge that comparative data cannot determine causality—how would households in Enduimet and Makame have fared without the wildlife management areas? Recent studies have shown promise in overcoming this issue of counterfactuals and identifying causal mechanisms of environmental initiatives (reviewed in Ferraro & Hanauer, 2014a). Although we evaluated our research questions using a controlled analysis and comparatively rich household-level data (Agrawal, 2014), we recognize the need for future research to take a more mechanistic approach to explain the processes underlying our observed outcomes.

In their current form wildlife management areas are not a cure-all for trade-offs between biodiversity and livelihoods in protected area landscapes in Tanzania. However, our findings indicate that wildlife management areas specifically, and community-based conservation more generally, can be a promising approach as long as policy-makers guide strategies following

evidence of project successes and failures (Tallis et al., 2008), with close attention to variability in project performance (Miteva et al., 2012), and the specific mechanisms through which outcomes are achieved (Agrawal & Redford, 2006). Evidence for variable outcomes should stimulate research that identifies the best fit between strategy and context, using larger samples of well-controlled data (Borgerhoff Mulder & Coppolillo, 2005). In the meantime, quantitative research such as ours on the effectiveness of community-based conservation must be comfortable with small effect sizes and substantial variation. Similar approaches applied across the tropics will promote improved strategies and management, and ultimately better conservation outcomes.

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2.7. Figures

Fig. 2.1. Study site in northern Tanzania.
The locations of Makao, Burunge, Enduimet and Makame wildlife management areas (WMA) and the 40 study villages in (CA, conservation area; GR, game reserve; NP, National Park).

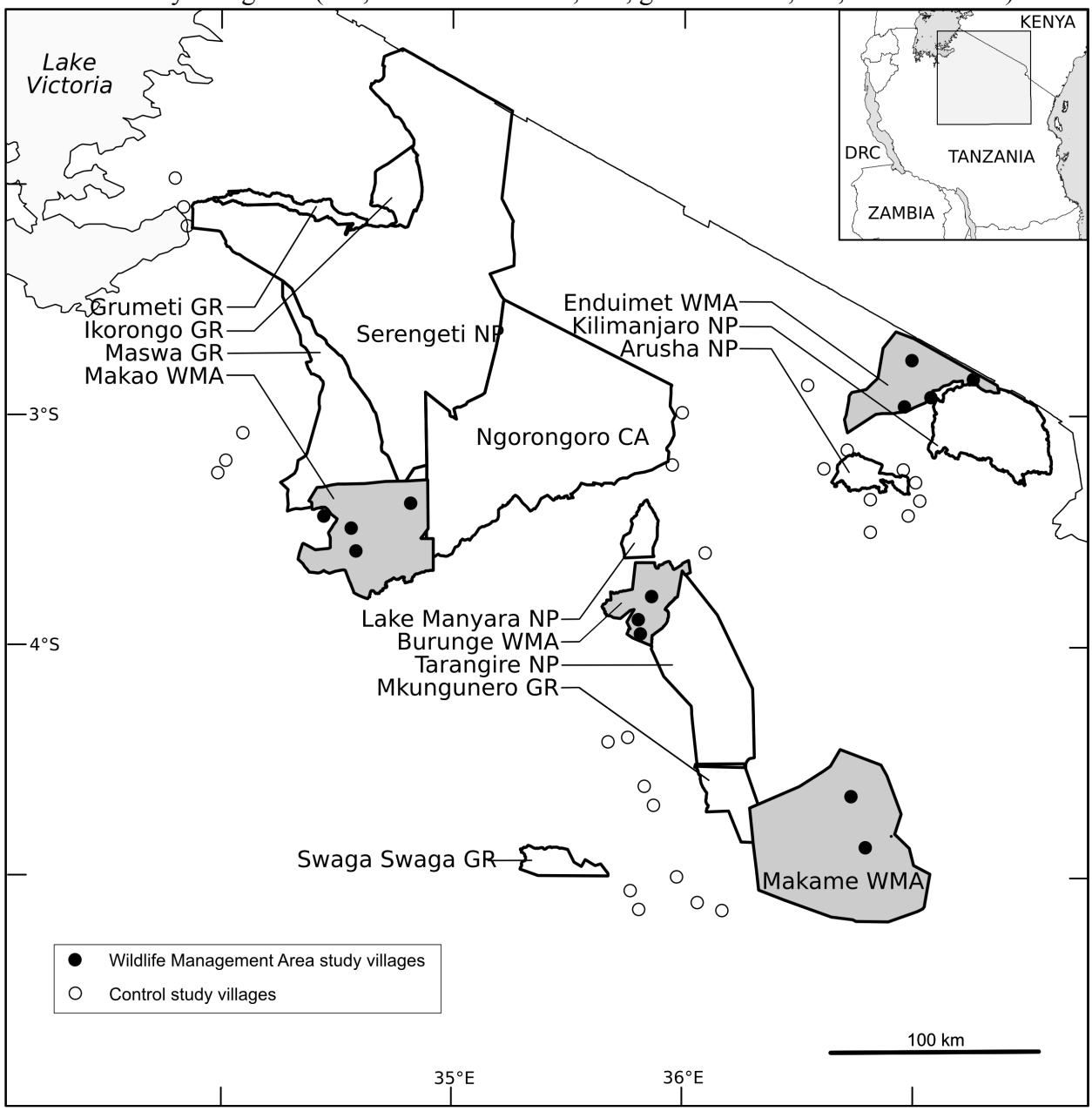


Fig. 2.2. Posterior densities of the effects of wildlife management areas on household food insecurity.
The main effect estimate is shared by all households within a wildlife management area and may be interpreted as the mean effect on the odds of experiencing more severe food insecurity. Varying effects make unique adjustments for each wildlife management area; the estimates displayed for each wildlife management area are the additive results of the main effect plus these unique adjustments.

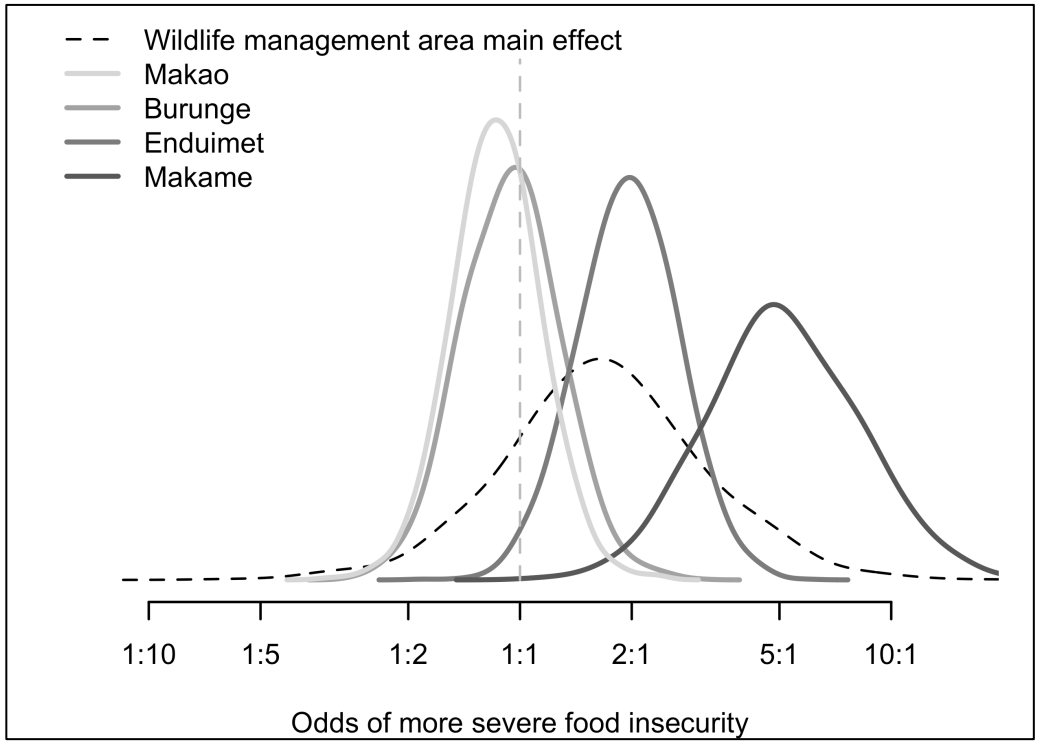
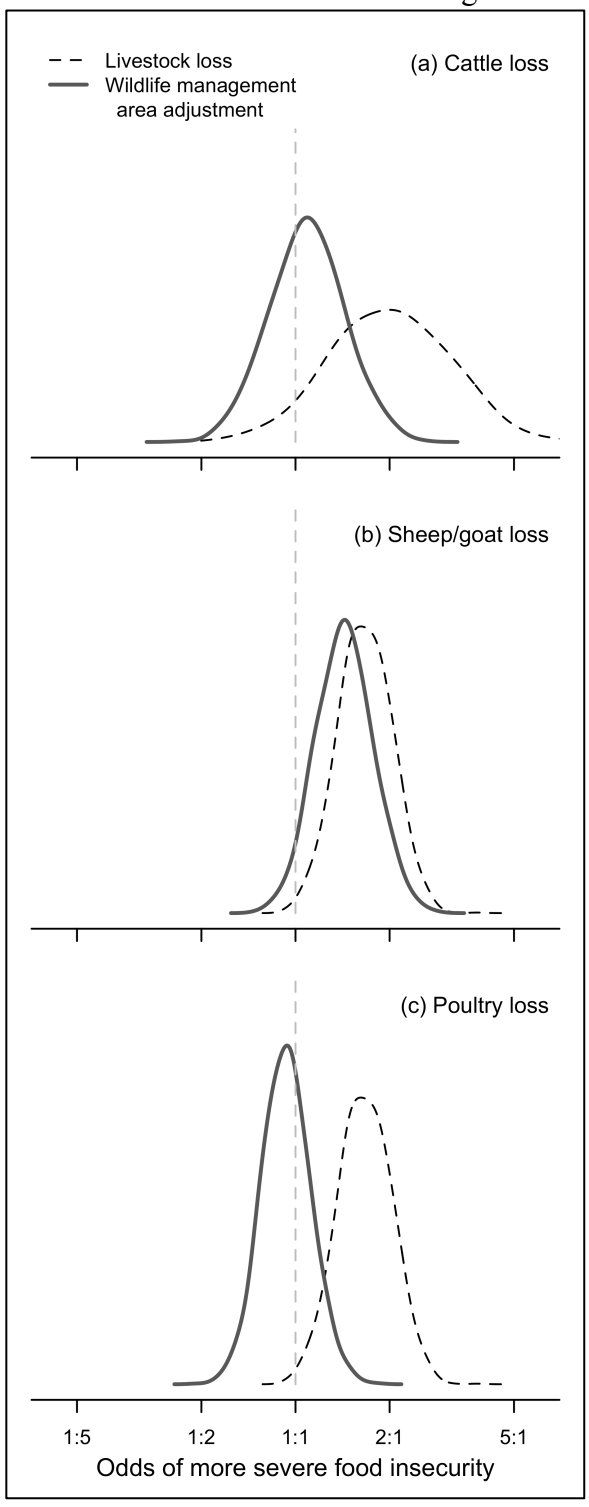


Fig. 2.3. Posterior densities of the effects of human–wildlife conflict on household food insecurity.

Main effects of livestock loss (dashed lines) are shared by all households losing (a) cattle, (b) sheep or goats, and (c) poultry to wildlife. Adjustments are made for households that also participate in a wildlife management area, and are the additive result of the main effect of livestock loss and the wildlife management area interaction effect.



2.8. Tables

Table 2.1: Characteristics of the survey samples.
Values based on wildlife management area status, with number of villages, number of households, proportion of households categorized as severely food insecure, mean household wealth index (based on ownership of items, excluding livestock and land; Supplementary Material 2), and proportion of households that experienced losses of livestock, cattle, sheep & goats, poultry or crops as a result of human-wildlife conflict in the 12 months prior to the survey.

	No. Of villages	No. of households	Food insecurity	Wealth index	Proportion of households that experienced human–wildlife conflict				
					Livestock loss[1]	Cattle loss	Goat loss	Poultry loss	Crop loss[2]
Non WMA	27	1723	0.44	1.50	0.34	0.02	0.10	0.27	0.18
WMA	13	776	0.55	0.98	0.54	0.19	0.33	0.28	0.47

2.9. Supplementary Information

Table S1. Numbers of villages and households represented in wildlife management areas (WMAs) and control areas.

Wildlife management area	No. of villages	No. of households
Burunge	3	188
Enduimet	4	236
Makame	2	112
Makao	4	240
Total (wildlife management area)	13	776
Total (control)	27	1,723

Table S2. Descriptive measures of households in the WMAs of Burunge, Enduimet, Makame and Makao, and in the control sample: the proportion of households categorized as severely food insecure, mean household wealth index (Supplementary Material 2); mean household productive livelihood asset holdings (cattle, sheep/goats, poultry, land); and proportion of households that experienced one or more incidents of human–wildlife conflict (loss of livestock, cattle, sheep/goats, poultry, crops) in the 12 months prior to the survey.

WMA	Severe food insecure	Wealth index	Productive livelihood asset holdings				Experience-of human–wildlife conflict				
			Cattle	Sheep, goats	Poultry	Land (ac)	Livestock	Cows	Sheep, goats	Poultry	Crops
Burunge	0.38	1.50	3.98	11.50	3.60	4.52	0.39	0.02	0.10	0.34	0.48
Enduimet	0.75	0.57	4.77	13.60	0.94	1.79	0.67	0.36	0.60	0.10	0.50
Makame	0.88	0.63	10.80	14.00	1.30	4.57	0.62	0.46	0.39	0.15	0.25
Makao	0.34	1.20	14.80	15.10	6.00	8.25	0.51	0.05	0.21	0.46	0.53
Control	0.44	1.50	3.48	7.18	4.60	3.91	0.34	0.02	0.10	0.27	0.10*

*Based on only 14 of the 27 control villages (Supplementary Material)

Figure S1. Predicted relationships between productive livelihood assets and the log-odds of food insecurity, with 95% credibility bands. Observed values for productive livelihood assets are plotted above the x-axis. Estimates are from the top-ranked model with controls (m1; Supplementary Material 3).

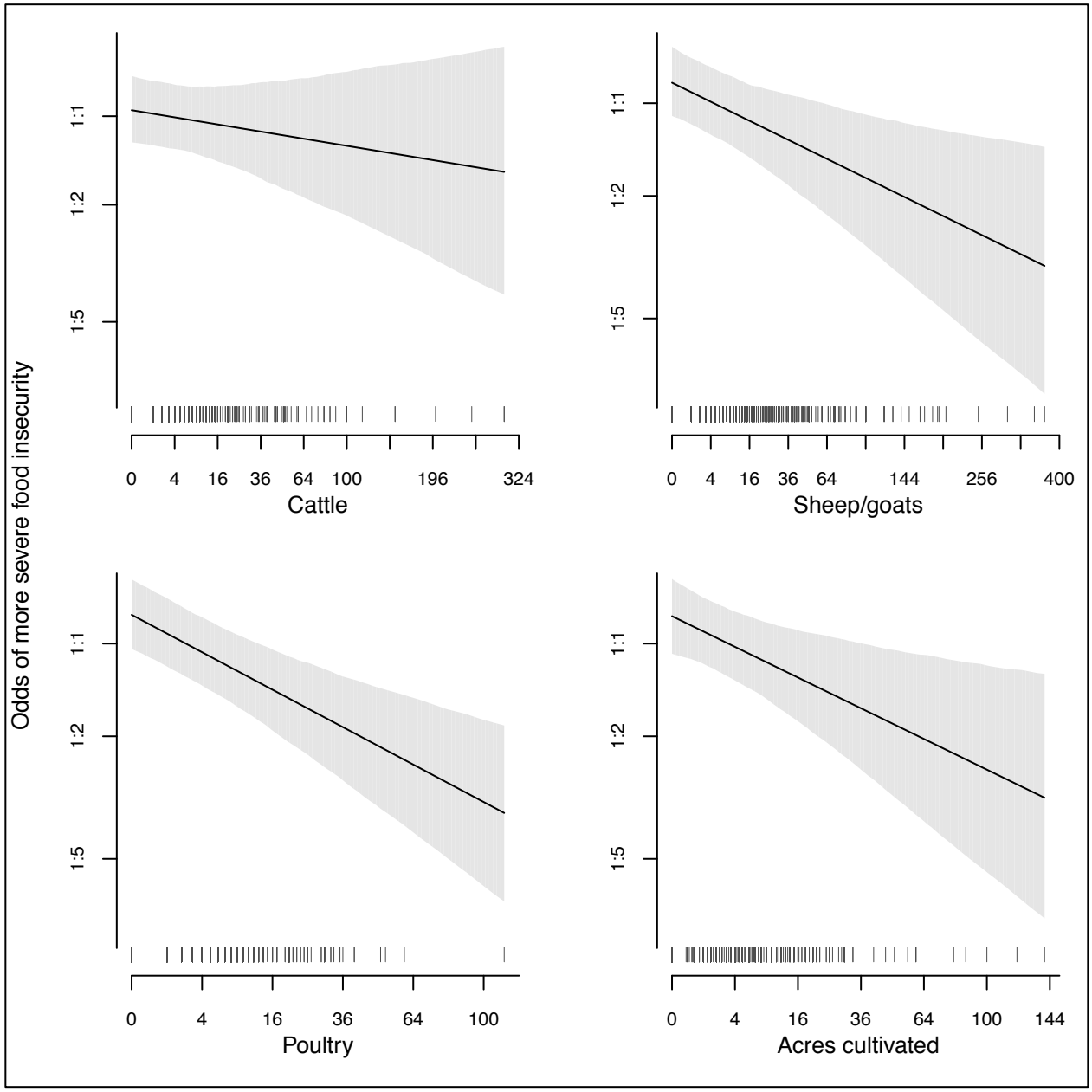
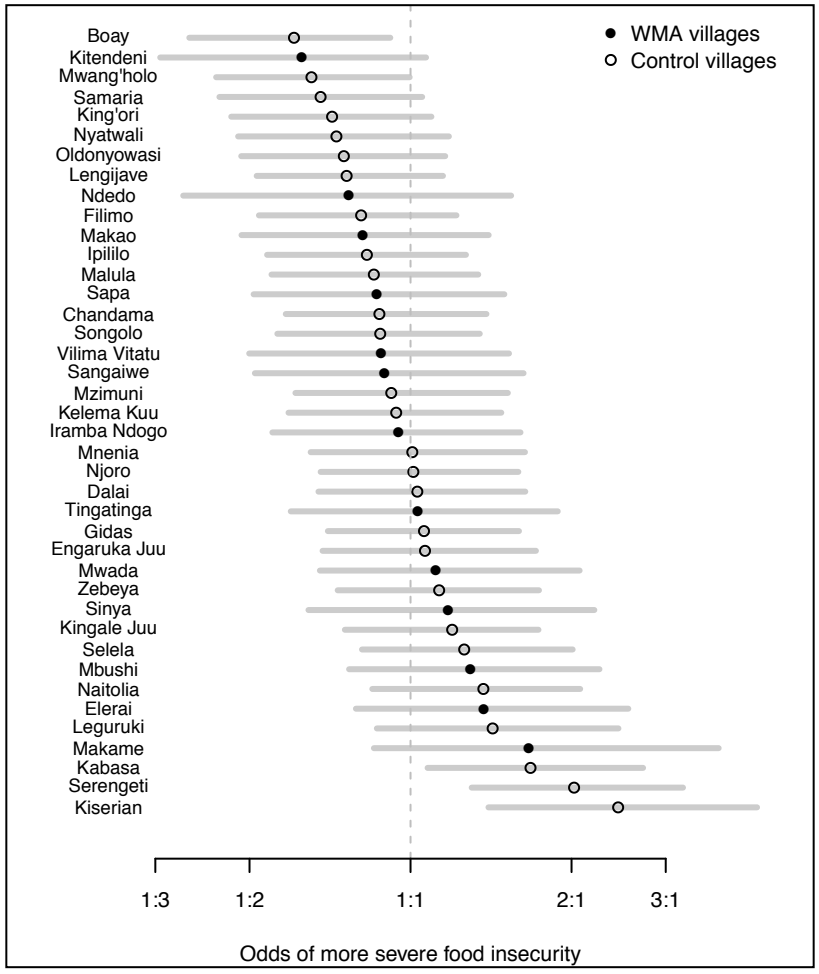


Figure S2. Village-level effects predicting household food insecurity. Varying intercept estimates for villages in wildlife management areas (WMAs) and control villages, with 95% CI, are listed from least to most food insecure, top to bottom. Estimates are from posterior densities of the top-ranked model with controls (m1; Supplementary Material 3).



Supplementary Material 1. Village site visits and household sample

Whole Village Project site visits and institutional assessments

The Whole Village Project was funded primarily by the U.S. Agency for International Development to provide baseline data with which to evaluate rural development projects. In each of the 56 villages surveyed, meetings were held with elected and appointed leaders. Activities

and objectives of the research were discussed and 60−75 households were selected at random from a complete list of residents. Village meetings also included a two-part institutional assessment. Firstly, leaders were asked about a number of village-level characteristics (e.g. recent in/out migration, religious composition, present facilities and social services). This part of the assessment included questions pertaining specifically to wildlife management areas and conservation efforts: communication of the village council with residents, recognized protected areas (including wildlife management areas) and their perceived costs and benefits, known village earnings from the protected areas, and known hunting and photography safari companies. Other informal interviews were completed (e.g. focus group interviews with the village health officer and female council members), but these were not considered in our study. Secondly, a group of randomly selected village residents were asked to list all active institutions and organizations in the village (including protected areas and wildlife management areas where applicable). Guided group discussion and debate (Chambers, 1992; Guijt & Pretty, 1992) produced numerical ratings of importance and effectiveness for each institution. For further information on the Whole Village Project, including site visit methodology and descriptive statistics, see Ritter et al. (2010) or contact MBM.

Village participation in wildlife management areas

Village participation in wildlife management areas is voluntary. However, external organizations such as conservation NGOs or private interests typically facilitate the establishment of wildlife management areas through the formation and registration of the Authorized Association. Land use plans and boundaries are proposed based on a number of factors and can include villages that

are not in favour of participation, either at the time of establishment or at a later date. Although each participating village typically contributes land to the wildlife areas, this is not always the case. Details of the participatory process are available in the report by Tetra Tech & Maliasili Initiatives (2013) and from the Government of Tanzania (2012).

Household sample

Field teams conducted structured questionnaires in each of the 60−75 households selected randomly during meetings with village leaders. Household heads responded to questionnaire modules surveying the cultural, demographic, nutritional and socioeconomic characteristics of household members. In our sample of 40 villages adjacent to protected areas, Whole Village Project technicians completed 2,571 household surveys (Ritter et al., 2010). Seventy-two of these records were incomplete and therefore deleted (Supplementary Material 3).

Supplementary Material 2. Household-level variables

Food insecurity

Food insecurity was measured using the household food insecurity access scale. To produce the scale the Whole Village Project incorporated nine questions (assessing the extent to which households experienced problems accessing food during the previous 30 days) into one of the household survey modules. Responses to these questions were used to assign each household to one of four categories of food security. For details on the household food insecurity access scale

and its implementation and validation see Coates et al. (2007) and, as applied in Tanzania, Knueppel et al. (2010).

Wealth index

Table 1 and Supplementary Table S2 report a wealth index based on purchasable household items, excluding livestock and land holdings. This index was calculated by ordinary principal components analysis of 37 binary variables applied to household data from the first 14 villages visited (see also Lawson et al., 2014). We used this index only for descriptive purposes. In our modelling framework we omitted the index and instead selected a discrete set of controls (see below).

Crop loss

Table 1 and Supplementary Table S2 report data on crop loss, which exist for only a subset of households. Questions pertaining to crop loss were omitted from surveys in 13 villages; this occurred in a particular phase of field sampling and exclusively in control villages. Because omissions were systematic or patterned in nature we did not impute missing values or include crop loss in our models. Where crop loss is reported in summary tables, means were calculated from the village subset where data existed.

Productive livelihood assets

The majority of sample households engaged in mixed livelihood strategies; for example, traditionally pastoralist Maasai account for the largest ethnic group in our sample, yet approximately one-third of Maasai households identified farming as their primary livelihood activity. The effects of cultivated land and livestock holdings on food insecurity and overall household well-being may vary between households; for example, whereas additional cattle may strongly predict lower food insecurity for a poor livestock keeper without cultivated land, similar beneficial effects may not be evident in wealthy farming households that invest cash crop profits in larger herds. Although we endeavoured to account for the sources of variation that explain food insecurity, both at the household and village levels, we acknowledge the potential confound of differential livelihood asset effects.

Control covariates included in models

A single set of household-level controls was selected to account for variation in wealth not captured by productive livelihood assets (livestock holdings and cultivated land). From variables recorded in the household surveys, we selected items that commonly indicate wealth accumulation in the study area. These were added to the basic model (Supplementary Material 3) and the most appropriate set was determined by model comparison of log-conditional predictive ordinates (logCPOs): bicycle, construction material of house (floor, walls and roofing), furnishings, mobile phone, motorbike, radio, sewing machine, and solar panel. This set was then included in all models.

Supplementary Material 3. Modelling strategy

Multilevel models

We refer to our approach throughout as multi-level or varying effects modelling (it is also known as hierarchical, random effects, or mixed effects modelling), inclusive of varying intercept and predictor effects. All models were fit using JAGS in R (Plummer, 2012; R Development Core Team, 2013). For simplicity, the main text specifies only the linear predictor μ of the basic model, a proportional-odds ordered logistic regression (Jackman, 2009). Here we specify the same model (m0), showing the cumulative logit link along with the linear predictor:

$$\Pr(y_{h,V,W} \leq c) = F(\tau_c - \mu_{h,V,W}),$$

We introduced focal predictors for livestock losses and their interactions with wildlife management area status, productive livelihood assets, and household level control covariates, aiming to build the simplest model (m1) that addressed our research questions. We then considered elaborations of m1 including, for example, additional interactions as well as varying predictor effects (m2–m4). All models containing 'livestock loss' included all three types of loss (cattle, sheep/goats, poultry), and those containing 'productive livelihood assets' included all four assets (cattle, sheep/goats, poultry, cultivated acres). All models except m0 contained an identical set of household-level controls (Supplementary Material 2).

Table S3 Varying effects, household-level effects, and logCPO values included in the linear predictor of models m0–m4.

Model	Varying effects	Household-level fixed effects	logCPO
m0	Village & wildlife management area intercepts	Wildlife management area status	–2,602
m1	Village & wildlife management area intercepts	Wildlife management area status, livestock loss, wildlife management area *loss, productive livelihood assets	–2,509
m2	Village & wildlife management area intercepts; livestock loss predictors by wildlife management area	Wildlife management area status, livestock loss, wildlife management area *loss, productive livelihood assets	–2,511
m3	Village & wildlife management area intercepts	Wildlife management area status, livestock loss, wildlife management area *loss, productive livelihood assets, wildlife management area *assets	–2,513
m4	Village & wildlife management area intercepts; livelihood asset predictors by wildlife management area	Wildlife management area status, livestock loss, wildlife management area *loss, productive livelihood assets	–2,512

We specified Gaussian priors with mean zero and variance 25 for fixed effects coefficients in the ordered logit model. The cut points $\tau_1<\tau_2<\tau_3$ were obtained by sorting independent Gaussian variables as in Jackman (2009). We specified Gaussian priors with mean zero and variance 100 for fixed effects coefficients in the Tobit model for imputation of acres cultivated (see below). We specified a uniform (0, 100) prior on the standard deviation of the Tobit outcome before truncation. All village and wildlife management area-level varying effects (both intercepts and slopes) were sampled from Gaussian distributions with mean zero and a uniform (0, 2) prior on the standard deviation.

Log-CPOs measure model quality by a cross-validation criterion (Gelfand, 1996) and are always negative; those closer to zero indicate more preferred models. Model m1 improves notably on m0 and is the best in the set of models we considered, although the more complex models, m2−m4, are only slightly less preferred than m1. As m1 is the simplest model among those

including the focal predictors, as well as the top-ranked model, the results presented in the main text are based on m1 unless otherwise stated.

Village-level varying intercepts and variation in food insecurity

Village-level effects from model m1 adjust for variation in baseline food insecurity (apart from that explained by varying wildlife management area and household fixed effects) between villages. There is no apparent pattern in village-level effects when comparing wildlife management area and control villages (Fig. S2); we understand this to imply that variation attributable to wildlife management area-specific factors have has been captured adequately by the wildlife management area effects of model m1. Food insecurity in control villages covers the sample range, and several control villages are noticeably food insecure, suggesting that our control sample is sufficiently variable. However, village-level effects are relatively small compared to the wildlife management area-level varying intercepts (the adjustments specific to unique wildlife management areas; Fig. 2) and household-level fixed effects (e.g. human−wildlife conflict, productive livelihood assets), and therefore villages appear to be less important sources of variation in food insecurity than household and wildlife management area-specific factors.

Missing data

Of the original dataset of 2,571 household surveys (Supplementary Material 1) 70 households were omitted because food insecurity, our outcome variable, was missing; two additional

households were omitted because many of the other key variables were missing. The dataset for model fitting thus comprised 2,499 households. The productive livelihood asset variable cultivated acres was missing in 111 of these 2,499 household records, although there was no apparent pattern that indicated systematic omission. We therefore assumed an ignorable missing-data mechanism (Little & Rubin, 2002, Chapter 10) and included stochastic imputation of cultivated acres as part of Markov chain Monte Carlo estimation of m1−m4. Observed values of cultivated acres vary continuously and are necessarily greater than or equal to zero (with many zeros in the sample), so cultivated acres can be treated as a limited dependent variable (Tobin, 1958). We therefore used a Tobit regression model (with village and wildlife management area-level varying intercepts and the same household-level predictors and controls included in the main models) to impute missing values of cultivated acres at each Markov chain Monte Carlo iteration (Plummer, 2011). We then carried these imputed values forward in m1−m4. We understand the resulting estimates in m1−m4 for focal predictors (such as wildlife management area status and livestock loss) to be averaged over the possible values of cultivated acres in cases where this variable was missing.

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CHAPTER 3: MIGRATION DECISION-MAKING IN A FRONTIER LANDSCAPE

3.1. Introduction

Agricultural Rural farmers and livestock keepers in developing countries make migration decisions as part of complex livelihood strategies. Recent scholarship advances our understanding of migration in and around natural resource and biodiversity areas specifically (Scholte & de Groot, 2010; Hoffman et al., 2011), and in the context of changing climate and environments more generally (Adamo & Izazola, 2010; Black et al., 2011; Adger et al., 2015; Hunter et al., 2015). Conceptual models predict migration behaviour from how environmental change differentially shapes household livelihoods (e.g., Black et al., 2011). For example, inadequate farmland or insufficient rainfall causes some households leave for various destinations while others remain, depending on capabilities and perceptions. Migration decisions are largely driven by access to productive natural resources and economic assets, and while social capital and local community ties may moderate decisions their role is viewed as less causal (Curran, 2002; Doevenspeck, 2011; Adams & Adger, 2013).

This study directly addresses three specific and less-studied features of migration-environment dynamics, namely the importance of social factors relative to environment and economy (de Haas, 2010), an explicit consideration of immobility or the decision to stay (Hunter et al., 2015), and linking migration decision-making with ongoing changes in frontier communities (Barbieri et al., 2009; Lopez-Carr, 2012). I focus on the transitional frontier, defined here to encompass sparsely populated forests and grasslands along with areas undergoing rapid deforestation and clearing to become human-dominated rural landscapes.

Close examination of migration-environment dynamics uncovers diverse and context-specific factors affecting migration decisions as an adaptive response (Black et al., 2011; McLeman, 2014; Warner & Afifi, 2014). For example, declining environmental conditions (e.g., available land, soil fertility, rainfall) often trigger decisions to migrate, and the response may be patterned by access to alternative local natural resources (Hunter et al., 2013), household wealth (Warner & Afifi, 2014), and social networks supported by kin in origin and destination areas (Curran, 2002) or by past migration experience (Gray, 2009). Empirical focus has been largely directed at rural-urban or labour migration and international migration, while relatively little research examines permanent migration within the frontier, especially from origin areas as the frontier is increasingly settled (Carr, 2008; Foresight, 2011). Moreover, even as deterministic land- and climate-driven models of migration are rethought, the barriers to migration or immobility remain understudied (Warner & Afifi, 2014; Adams, 2015), along with the implications of immobility for resource institutions and governance in rural communities (Charnley, 1997; Curran, 2002). Testing more comprehensive models is necessary to determine why frontier mobility persists and how households make decisions to move or remain based on perceptions of environmental, economic, and social conditions (Hunter et al., 2015).

Continued migration to and within frontier areas poses an environmental challenge, most directly in frontier landscapes through deforestation and land clearing for agriculture (Galaty, 1988; Painter & Durham, 1995; Brondizio & Moran, 2008). Where significant areas of the landscape are designated for biodiversity protection, migration can contribute to increasing pop densities at protected area borders and consequent isolation effects of protected ecosystems (DeFries et al., 2005; Joppa et al., 2009; Estes et al., 2012; Salerno et al., 2014). Furthermore, migration into frontier communities increases local demand on environmental resources such as shared forests

and pastures. These changes are understood to negatively impact local resource management institutions by weakening social bonds and diluting knowledge of rules and resource use practices (Charnley, 1997; Katz, 2000; Ostrom, 2000), though not universally (see Bauer, 1987; Atran et al., 1999).

As frontier areas transition into human-dominated agricultural landscapes, migration may continue into further, less-settled areas of the forest or grassland frontier (Galaty, 1988; Laurian et al., 1998; Katz, 2000). The environmental outcomes of in-migration and population growth in such areas are well documented (see above). However, there is a gap in forward-looking research linking migration decision-making with how migrant-resident communities change, for example, as length of residence increases and time horizons shift (Holmes, 2005), which has significant implications for managed natural resources and protected biodiversity (Curran, 2002; Pretty & Smith, 2004; Hartter et al., 2015).

This letter investigates past and future migration decisions in the transitional frontier landscape of western Tanzania (figure 1). It focuses on Sukuma agropastoralists, whose persistent expansion is associated with widespread environmental change. Agropastoralist mobility is largely explained simply by access to available land to maintain extensive farming and livestock keeping practices (Malcolm, 1953; Galaty, 1988; Brockington, 2001). I test this assumption by considering livelihood decisions as the result of a diverse set of resources and assets (Chambers & Conway, 1992; Ellis, 2000), and by building on a more comprehensive model of mobility from the migration-environment literature (e.g., Black et al., 2011). I ask two related questions about the drivers of rural migration using quantitative household surveys and observational data: (1) which factors drove past decisions to move, and (2) how do multiple factors differentially contribute to households' expectations of future out-migration from their present location?

Results uncover surprising features of migration decision-making and illustrate how investigating mobility can contribute to understanding natural resource conservation and management in the frontier.

3.2. Materials and methods

3.2.1 Livelihoods and migration in rural Tanzania

The livelihoods framework conceptualizes household decision-making through adaptation or diversification strategies given available capital assets (Chambers & Conway, 1992). It has been effectively applied to explain migration decisions in varying contexts (de Sherbinin et al., 2008; Black et al., 2011). For example, households may utilize migration to access capital assets that are unavailable in their current location, such as moving livestock to non-local, productive grazing areas or sending household members to access wage labour opportunities. Migration is costly, and decisions to move may be facilitated by existing forms of capital such as cash reserves, local family ties, or kin networks present destination areas.

The livelihoods framework provides an integrated approach to investigating individual and household behaviours in rural developing country contexts (Ellis, 2000), while recognizing broader sociopolitical and biophysical processes that shape the value of household assets (Scoones, 2009). The agropastoralist study population engages in a limited set of livelihood activities (e.g., few households engage in or hire wage labour, few possess physical assets such as tractors, few hold bank accounts or access credit). Therefore, as a simplifying step this study categorizes livelihood capital factors predicting migration as access to environmental resources, social capital, and household productive assets.

Frontier migration of agropastoralists in Tanzania has been associated with large-scale environmental change since the mid-20th Century (Malcolm, 1953; Brandstrom, 1985; Salerno et al., Submitted). Continued expansion into the frontier results in extensive forest clearing, overgrazing, soil degradation, and social conflict in destination communities (Charnley, 1997; Brockington, 2001; Borgerhoff Mulder et al., 2007). This study focuses on the agropastoralist Sukuma, the largest ethnic group in Tanzania, whose mobility is characterized by multiple, successive migration events over time (Galaty, 1988). Households typically settle in a frontier area, clear land for rainfed agriculture and pasture, and then migrate again when population densities limit agricultural expansion and central-place livestock keeping practices (Coppolillo, 2001).

Mobility patterns result in a transitional frontier of increasing human and livestock densities and decreasing land and natural resource availability. Households migrate as a single unit, rarely sending temporary labour migrants, however young men often move to select and clear a site before their family group follows. Sukuma society is exceptionally cooperative and dependent on a complex social hierarchy linking distant villages through clan membership under one chiefdom (Abrahams, 1967; Paciotti et al., 2005). Sukuma mobility is described as causing a 'cascade effect' of slow-onset environmental change (Charnley, 1997) over the expanding transitional frontier.

3.2.2 Study system, data collection, and analysis analyses

Large-scale migration of Sukuma agropastoralists to the Katavi-Rukwa study system (figure 1) began in the 1970's following repeated droughts in their origin areas of northern and central Tanzania (Brockington, 2001). Katavi Region currently experiences the highest rural population

growth rate in the nation (URT, 2013). The study system was selected because it includes mixed migrant-resident communities experiencing varying degrees of both in- and out-migration, along with land clearing and resource use pressures (Borgerhoff Mulder et al., 2007). Study communities are located in a landscape with a diversity of protected area types and ongoing community-based natural resource management initiatives. Katavi National Park and adjacent strict protected areas are of critical conservation importance. Forest reserves and village resource areas function as buffers to the core biodiversity areas as well as constitute productive managed lands for timber, fuelwood, and grazing (Pailler et al., 2015; Salerno et al., 2015). Prior to the mid-1900s, the resident population included Pimbwe, Fipa, Bende, and Konongo ethnic groups at relatively low population densities (Willis, 1966). While state authorities and international donors recognize the vast extent of agropastoralist expansion, it is the local village governments that are most directly involved in the struggle to manage forests, farmland, and pastures, often without support from higher administrative levels (Salerno et al., Submitted).

Fieldwork was conducted between 2011 and 2013 in seven villages bordering Katavi National Park and adjacent conservation areas. Study villages were identified through interviews with region and district officials during pilot work in 2011. Villages were selected in the major areas of in-migrant settlement within the three distinct ethnic and agroecological areas bordering the Park. In consultation with village officials, one to three sub-villages were selected for household surveys, which were conducted in 2012 and 2013. Sample households were randomly selected from sub-village rosters, and elected sub-village chairmen provided introductions at each household prior to surveys. Household compounds are typically comprised of the household head, married and unmarried sons, and wives and children. Compounds are located away from village centres, and it is not uncommon for them to include 30 or more family members and

hundreds of livestock. Following verbal consent, surveys were conducted with household heads in Kiswahili or translated to Kisukuma when necessary.

Household surveys yielded qualitative migration narratives along with two quantitative migration outcome measures and household-level predictor variables. Household heads were asked to recall their family migration histories, which were recorded as narratives. Next, household heads were asked to free-list and rank the primary factors they thought were important in driving past migration decisions (number of reasons stated ranged from 1-5, mode=3; for similar methods see Baird et al., 2009; Doevenspeck, 2011). Responses were coded through emic categories: agriculture (e.g., land for farming, pasture for grazing), environment (e.g., rainfall, climate), family-community (e.g., proximity to kin, peaceful community, household health), business (e.g., access to roads, cash crop markets) and development (e.g., health centres, schools).

Respondents then quantified their expectation of future migration through a simple activity. Respondents were given a set number of maize kernels and, following multiple example rounds, asked to demonstrate the chances or likelihood that they would move their family to an area outside of their current ward at any point in the future by placing a fraction of their maize in a pile. This method was adapted from experimental economics as an effective tool for measuring respondent perceptions of probability or expectation of future events and was effective in the low literacy study population of Katavi-Rukwa (Luseno et al., 2003; Delavande et al., 2011; McKenzie et al., 2013). While acknowledging that this outcome variable measures only perceptions of future migration behaviours, these perceptions reflect the cognitive process through which certain households self-select to make the decision to migrate (Curran, 2002; de Haas, 2010; Kyle & Koikkalainen, 2011; Adams & Adger, 2013).

In addition, quantitative surveys measured predictor variables to include in statistical models. Variables are organized into categories based on simplified groupings of livelihood capital assets (table 1): access to environmental resources (farm productivity, land scarcity, conservation area benefits, conservation area costs, conservation area knowledge), social capital (local community kin ties, participation in local community leadership, non-local livelihood sharing, non-local kin networks, mobility networks, migration experience, years resident), and household productive assets (land owned, cattle owned, age, education, number of wives, number of sons). While variables can be categorized in different ways, and additional categories can be applied (e.g., financial, human, physical capitals), the steps taken here are appropriate to agropastoralist livelihoods in the study system and used primarily as a meaningful organizational tool.

A single, multilevel regression model was fitted to test the associations between expected out-migration and the suite of predictor variables. The model estimated respondent expectations of migration (indicated by the total kernels of maize placed out of 20) as a binomial outcome using Markov-chain Monte Carlo methods (Plummer, 2003; for model structure and prior specification see Gelman & Hill, 2007)p381,430. Due to the hierarchical structure of the data—households clustered in sub-villages and sub-villages in villages—the model included varying intercept (i.e., random) effects for both village and sub-village. These effects control for unobserved differences in the outcome variable shared by households within different levels of clustering. Varying intercept effects also allow for examination of how the variance in out-migration behaviour is partitioned between these levels, which explains, for example, whether single villages are composed of heterogeneous households or whether more important differences in migration behaviour exist between villages.

The two related research questions—(1) which factors drove past decisions to move and (2) which factors contribute to future migration expectations—are evaluated through different approaches. The relative importance of different reasons for past migration (Question 1) is assessed through descriptive statistics of the emic categories stated above. The influence of environmental, social, and productive assets on anticipated out-migration (Question 2) is evaluated through coefficient estimates from the statistical model. These estimates are interpreted using graphs displaying their relative size and uncertainty, as well as by calculating their estimated multiplicative effect on the odds of anticipated out-migration (all estimates reported with a 95% credibility interval). As argued in the Introduction, because of the importance of understanding migration decisions in terms of future changes in environments and communities, I focus primarily on findings of the statistical model.

3.3. Results

In response to the question, "What reasons were important in driving your family's past migration decisions?" (Question 1), land for agriculture was most often mentioned (figure 2a) and most highly ranked (figure 2b) by household heads. Within the agriculture category, when respondents chose to elaborate they cited farmland as more important and cited it twice as frequently as grazing land. Environmental reasons ranked second and were predominantly related to rainfall. Specific reasons related to family included a cooperative and peaceful community, household health, and food security. Reasons related to business and development referenced access to cash crop markets and services such as schools and roads. Though 66% of respondents knew of conservation areas and 14% knew of conservation outreach activities prior to arriving, only two individuals cited conservation as an important reason driving their past migration decisions. One respondent moved after being evicted from a national park, while

another moved to gain access to the village forest reserve (these reasons are included in the Environment category in Figure 2).

Estimates from the multilevel model demonstrate the relative strength of environmental, social, and productive economic factors in predicting anticipated out-migration (Question 2; figure 3). These estimates suggest that social capital assets have the strongest associations with out-migration. Approximately one-quarter of households maintained strong local community kin ties or leadership roles (22% and 26%, respectively). The presence of local kin decreases the odds of anticipated out-migration by a factor of 0.74 (credibility interval: [0.60, 0.90], and participation in local leadership decreases the odds by 0.80 [0.67, 0.95]. Conversely, the odds are increased by: non-local livelihood sharing (1.36 [1.15, 1.61]; e.g., cattle kept in other districts), non-local kin networks (1.06 [1.03, 1.09]), and previous migration following close kin (1.17 [1.00, 1.36]). Migration history has a less certain effect, such that additional past migration events increase odds by 1.05 [0.99, 1.12].

Certain predictors describing access to environmental resources are also consistently associated with anticipated out-migration (figure 3). All study communities were located adjacent to protected areas, and a substantial proportion of households experienced conservation-related benefits (45%) and demonstrated knowledge of conservation area rules and ongoing outreach activities (43%). Experiencing benefits and demonstrating knowledge decrease the odds of anticipated out-migration by 0.82 [0.71, 0.96] and 0.84 [0.71, 0.98], respectively. Respondents mentioned costs associated with conservation, such as restricted resource access or conflicts with wildlife, yet the effect of these costs is uncertain. Onfarm environmental resources are weakly predictive of anticipated out-migration: greater land scarcity increases odds by 1.02 [1.01, 1.04], while farm productivity has no consistent association.

Predictors describing household productive assets include agricultural assets and household characteristics (figure 3). Additional cattle holdings and years of age of household head decrease the odds of anticipated out-migration by a factor of 0.95 [0.92, 0.98] and 0.98 [0.98, 0.99], respectively. Farm size as well as education and number of sons and wives of the household head have no consistent associations with out-migration.

3.4. Discussion

Social capital is highly predictive of anticipated out-migration in the transitional frontier. Quantitative findings demonstrate the relative strength of different types of social capital on migration decisions in the presence of environmental and economic assets. For example, consider a dichotomy comparing two hypothetical households: model results imply that one household with assets such as cattle shared through networks in other areas, with kin identified across more regions, and with a history of following kin during previous migrations is nearly three times as likely to anticipate leaving their current location than an otherwise similar household that maintains stronger local kin ties and participates in local leadership roles. The estimation from the multilevel model also shows that environmental resources such as access to additional land and economic capital such as cattle holdings indeed influence migration expectations, yet their influence is relatively weak (figure 3). Households expect to maintain longer residence when they actively benefit from protected biodiversity and resource use areas and when they show greater knowledge of these areas and the rules governing their use. Furthermore, when compared to reasons given for past migration decisions, the importance of social factors influencing future expectations (as estimated by the model) suggests that migration decision-making may be changing.

The relative importance of social capital is surprising when considering the preponderance of recent research highlighting environmental factors and economic or productive factors as driving forces in migration decision-making (Black et al., 2011; Warner & Afifi, 2014). Moreover, findings from this letter go beyond showing that social capital matters (Gray, 2009) to demonstrate the distinct effects of engagement within migrants' present community (promoting immobility) in contrast to the maintenance of networks in previous and potential destination areas (promoting out-migration) (Massey, 1990; de Haas, 2010). Doevenspeck (2011) found similar strong effects of social capital compared to environmental factors on out-migration behaviours among migrants in rural Benin. Migrants maintained non-local social networks and engaged in multiple migrations following kin in destination communities. Results here add to these and other quantitative and qualitative evidence supporting the importance of social capital in promoting outmigration within rural areas (Barbieri et al., 2009; Massey et al., 2010), as well as local social ties and community engagement promoting continued residence (Carr, 2005; Warner, 2010). Longer term residence has implications for migrants shifting their time horizons, perhaps adopting more sustainable resource use practices, and supporting community institutions (Charnley, 1997; Katz, 2000; Holmes, 2005), which I address below.

Environmental resources in the forms of available and productive land affect national-scale migration patterns in Tanzania (Galaty, 1988; Charnley, 1997; Salerno et al., 2014), and access to land resources is indeed a principle factor shaping frontier migration elsewhere (Bilsborrow, 1987; Painter & Durham, 1995; Lopez-Carr, 2012). Quantitative results presented in this study indicate that land scarcity consistently predicts expected out-migration for households in Katavi, but the effect is relatively small, and while farm productivity may suggest longer residence the effect is weak and imprecise. These results support the more nuanced understanding of land in

the context of different forms of capital—clearly migrant farmers and cattle keepers consider land availability in potential frontier areas, but decision-making selects, and sometimes quite strongly for example, for those households with strong networks of kin already in the destination (Massey, 1990; Curran, 2002; Gray, 2009; de Haas, 2010).

Analyses considered household interactions with local protected areas as forms of environmental resources or natural capital. This study was designed so that all surveyed households were in close proximity to Katavi National Park or adjacent strict and community managed protected areas, which allowed for examination of the effect of these areas on migration decision-making. Strict and community protected areas are unlikely to attract migrants (Estes et al., 2012; Zommers & MacDonald, 2012; Hartter et al., 2014a; Salerno et al., 2014), rather patterns of observed migration to borders are likely the result of protected areas being disproportionately located in regions of low population density (Joppa & Pfaff, 2009; Scholte & de Groot, 2010). Analyses addressed this claim and showed through retrospective questioning that, while the majority of respondents knew of the ongoing conservation activities prior to arriving, neither conservation areas nor associated benefits factored into their stated reasons for migration to the Katavi study area.

However, results predicting future migration suggest that positive interactions with adjacent protected areas affect longer residence. Although variable, the effects of perceived benefit and greater knowledge of conservation areas and rules are quite strong. Benefits include fuelwood, timber, and grazing areas provided by local resource areas such as community forest reserves (Pailler et al., 2015). Many households also cited non-material ecosystem services of Katavi National Park, such as rainfall and keeping wild animals away from settlements, which may significantly shape positive attitudes regarding conservation (Holmes, 2003; Hartter et al.,

2014b). Together, resource use and biodiversity areas constitute natural capital and contribute to how households make adaptive decisions through migration (Adams & Adger, 2013; Hunter et al., 2013).

Interestingly, household economic or productive capital assets do not drive migration decisions in this study, although wealth in terms of cattle is weakly associated with longer residence. Sukuma agropastoralists invest nearly all cash crop profits in cattle, and cattle wealth has principally characterized their 'hypermobility' in the past (Malcolm, 1953; Brandstrom, 1985; Galaty, 1988). In terms of household assets considered here, it appears now that the younger, perhaps less-educated, and poorer households are moving, which may indicate vulnerability induced out-migration, but again these effects are relatively weak, and the data do not allow for an explicit test of vulnerability versus opportunity driven mobility. More generally, these findings support the growing consensus that environmental decline, as is ongoing in Katavi, precipitates different migrant streams making different decisions to both leave and remain based on accessible livelihood assets and resources (Gray, 2009; Massey et al., 2010; Black et al., 2011; Warner & Afifi, 2014).

Comparing results describing past and future migrations suggests that mobility decisions may be changing. Explicit statements by respondents demonstrate that available farmland was the primary driver of their past migration decisions (figure 2), and other factors such as available pastures, rainfall, and family health were important but secondary. These factors were not constrained into the same categories applied to predictors in the anticipated migration model (table 1), but past reasons clearly indicate that acquiring land (household productive assets) was viewed as more important than other environmental resources or social factors such as rainfall for crops or community relations. In contrast, the model demonstrates local social ties that

support longer residence for some, along with non-local social networks that promote mobility for others, play critical roles in anticipated future migration decisions (figure 3). Factors affecting past and likely future moves were measured using different methods. This was done primarily to allow respondents to generalize reasons from the past across their multiple migrations and elicit the most accurate responses. While acknowledging the different methods of questioning, the contrasting past and future reasons may be explained by the increasing population and changing environment of the transitional frontier.

Rural households in Tanzania have historically utilized mobility to avoid adopting more intensive livelihood practices (Galaty, 1988; Brockington, 2001) (see also Bilsborrow & Ogendo, 1992). Shorter time horizons are used to explain the environmentally destructive behaviours of in-migrants and their lack of cooperation and investment in natural resource management in frontier areas (Charnley, 1997; Katz, 2000; Holmes, 2005). Yet household heads in Katavi repeatedly stated during interviews that the land was filling up, that families were squeezed, and that many were changing their behaviours. That is, some households may view what was previously the unsettled frontier as having since transitioned into a more densely populated rural landscape. Such frontier areas as a destination may be therefore less attractive for extensive farming and grazing. Together the changing environmental resource conditions coinciding with changing behaviours suggest a potential counter-narrative to the hypermobile, "leapfrog" characterization of agropastoralist mobility (Galaty, 1988; Charnley, 1997; Brockington, 2001).

Finally, I briefly discuss the implications for changing mobility behaviours in terms of the environment and local institutions managing natural resources. Evidence presented in this letter suggests that longer-term residence is predicted by those who are socially embedded and also who benefit from and possess knowledge of local resources areas. Embeddedness is

characterized by the 26% of respondents holding leadership positions in the local community, such as with the village government or local rule-enforcing groups. If changing migration behaviours that promote longer residence indeed exist along with greater engagement in institutions, then such change presents a possible alternative to the widespread prediction that migration into rural areas negatively affects cooperative behaviours and natural resource management (Ostrom, 2000; Curran, 2002).

This study did not examine migrant resource use practices explicitly. However, findings demonstrate that necessary precursors may be evolving for improved natural resource management and conservation among migrants in this frontier area. In general, where communities themselves govern the use of farmland, pastures, and forests, individual participation in rule-making and enforcement is more likely to result in healthy resource systems supporting local livelihoods (Rustagi et al., 2010; Persha et al., 2011). Indeed, at the time of this research, in-migrant families were actively enforcing village forest reserve boundaries, including the investigation and apprehension of neighbouring villagers illegally harvesting timber (pers. comm. P. Genda; Genda, 2012). Recent evidence from the study site also suggests that migrant-led institutions may be combatting the illegal killing of lions in the areas in and around Katavi National Park (Fitzherbert et al., 2014). Future empirical efforts are necessary to explicitly examine how migrants affect local institutions (Curran, 2002) and whether these dynamics can help explain the potential emergence and persistence of cooperative behaviour for improved natural resource management (Waring et al., 2015).

In sum, this study contributes to a more comprehensive understanding of how rural people make adaptive decisions through mobility within changing frontier environments. Findings demonstrate the importance of migrant social capital in shaping future decisions, as well as how

protected natural resource areas can affect these future decisions. Findings also highlight the utility of novel field methods for quantifying migration decisions, which can support analyses to tease apart the differential effects of factors influencing mobility behaviours. Looking ahead, in frontier areas where mobility is associated with widespread environmental degradation, models of migration should investigate future decisions and the implications for changing migrant-resident communities. These approaches may be instrumental in understanding how communities adapt and how improved local management of natural resource systems can arise and persist.

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3.6. Figures

Fig. 3.1. Katavi-Rukwa study system, western Tanzania.
Sample households in seven study villages (red points) were visited 2012 – 2013. Villages are located in close proximity to strict protected area borders (blue polygons). Significant forest clearing is evident near the study villages (e.g., adjacent to the north-eastern villages) and in other areas adjacent to protected area borders. Study site extent: 6.110333°S – 7.647705°S, 30.484913°E – 32.790203°E. Data sources: IUCN & UNEP-WCMC 2014; Esri, DigitalGlobe.

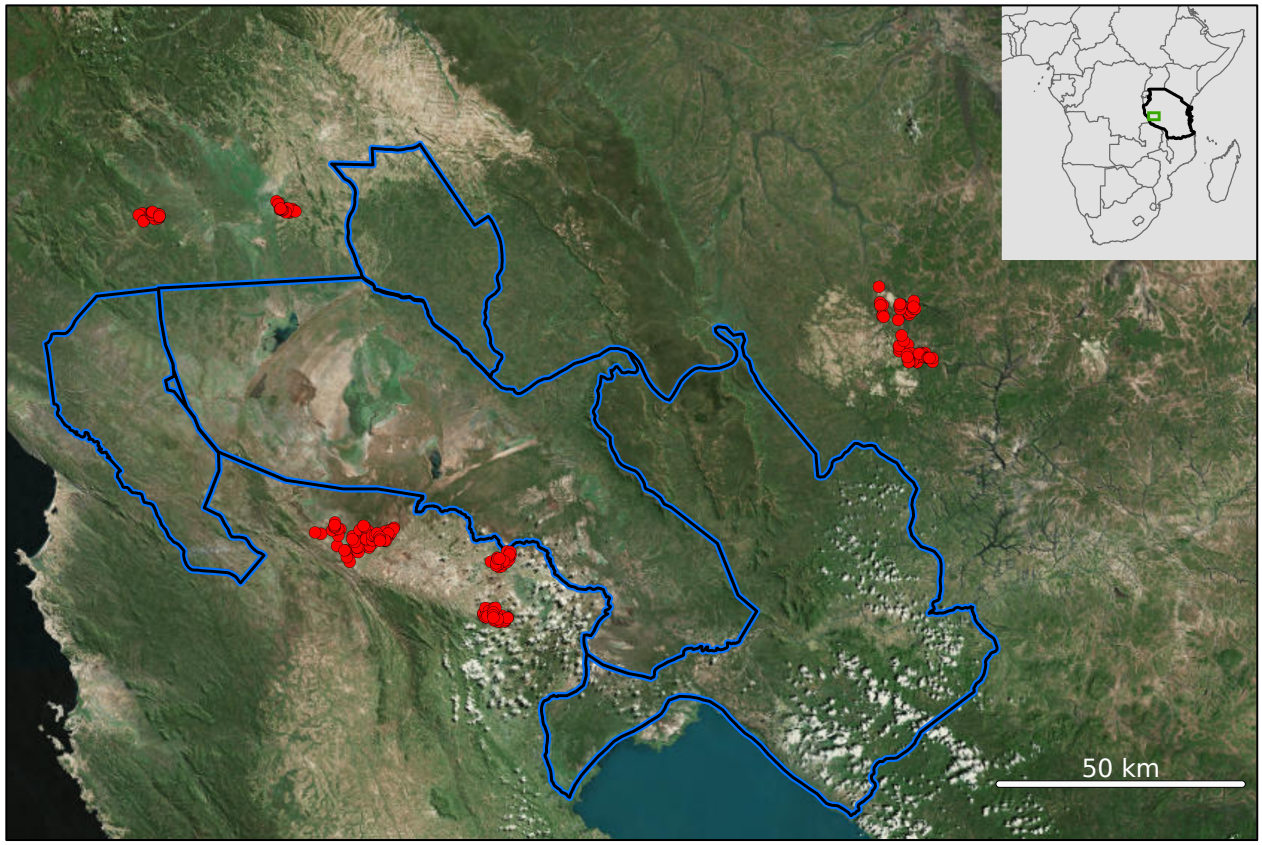


Fig. 3.2. Reasons stated by household heads as important in driving their pervious migration decisions.
Sum of all reasons listed (a); most important single reason stated (b). Respondents freelisted (i.e., stated without prompting) and ranked as many reasons as they chose appropriate (from 1-5, mode=3). Every respondent listed at least one reason, while only two respondents listed five reasons. Individual responses were coded into one of five emic categories (e.g., good soil for growing maize or free access to pastures were categorized as 'Agriculture', nearby primary schools as 'Development'); therefore, one respondent could list multiple reasons assigned to the 'Agriculture category'.

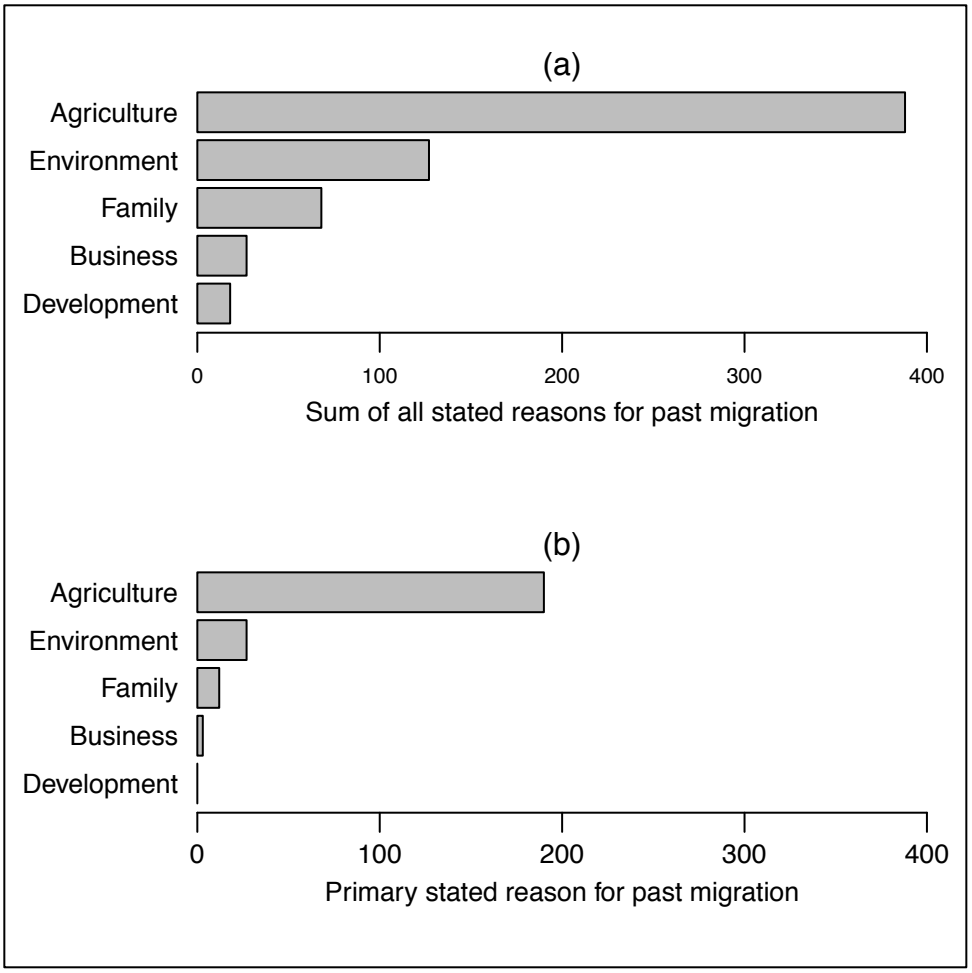
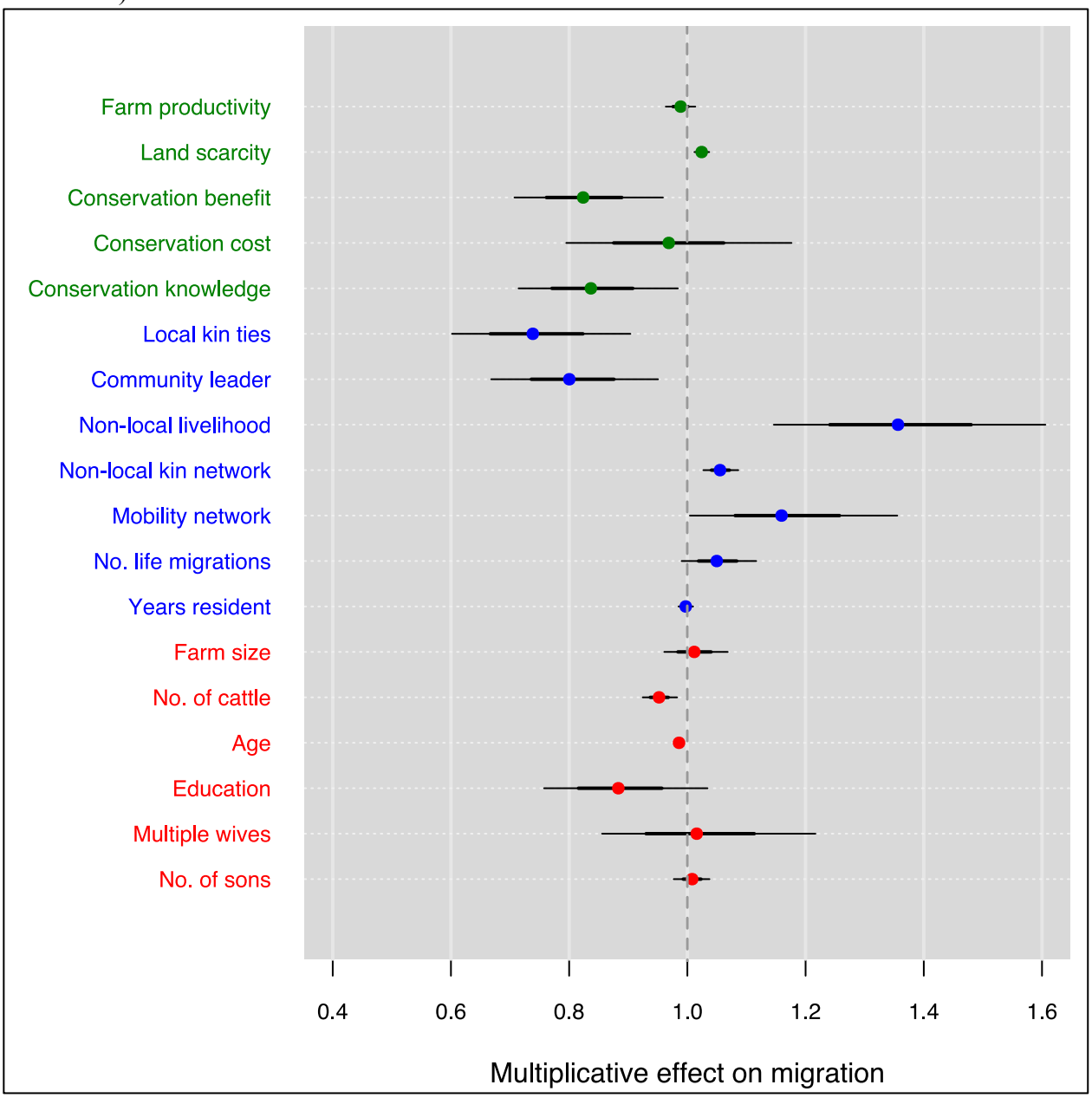


Fig. 3.3. Model estimates of main effects predicting anticipated out-migration.
Posterior mean estimates of all fixed effect coefficients from the multi-level model are displayed in terms of their multiplicative effect on the odds of out-migration, with 95% and 68% credibility intervals. Colours correspond to predictor variable categories (table 1): environmental resources (green), social capital (blue), household productive assets (red). On this scale, a coefficient estimate of 1 is equivalent to no net effect (i.e., the indifference value; dashed grey line). Each plotted estimate represents the additive effect on anticipated out-migration of a binary predictor variable or the unit change of a continuous or count predictor (see table 1 for variable definitions).



3.7. Tables

Table 3.1 Factors of migration decision-making.
Variables predicted to influence migration decisions are drawn from the literature and organized into the three general categories (column 1, left): environmental resources, social capital, household productive assets. All variables listed (columns 2 – 4) are included in the statistical model predicting anticipated out-migration. Sample means (col. 4) are reported on observed values (not transformed).

	Predictor variable	Variable description	Sample mean (SE)
Environmental resources	Farm productivity	Maize yield in sacks per acre; continuous	4.48 (0.21)
	Land scarcity	Number of years since farm was last expanded; discrete	8.23 (0.43)
	Conservation benefits	Perceived by household from protected areas or associated outreach activities; binary	0.45 (0.03)
	Conservation costs	Perceived by household from protected areas or associated outreach activities; binary	0.23 (0.03)
	Conservation knowledge	Stated protected area names and rules, location, and/or outreach project activities; binary	0.43 (0.03)
Social capital	Local community kin ties[a]	Presence of multiple kin households in current location; binary	0.23 (0.03)
	Local community leadership	Household head holds leadership role in local institution; binary	0.26 (0.03)
	Non-local livelihood sharing	Household shares farm, livestock, or business activities with non-local individuals; binary	0.24 (0.03)
	Non-local kin networks[a]	Number of regions where household identifies kin or close friends; discrete	6.023 (0.20)
	Mobility networks[a]	Household's most recent migration was facilitated by kin arriving first to the destination location; binary	0.65 (0.03)
	Total migrations	Number of migrations over the lifetime of the household head; discrete	2.53 (0.09)
	Years resident	Number of years in current location; discrete	10.69 (0.60)
Household productive assets	Farm size	Number of acres owned by household; discrete (sq. root transformed for binomial model)	16.70 (1.19)
	Cattle	Owned by household; discrete (sq. root transformed for binomial model)	24.65 (2.64)
	Age of household head	Discrete	47.91 (0.89)
	Education of household head	Attended more than 3yrs primary school; binary	0.56 (0.03)
	Multiple wives	Married to household head; binary	0.38 (0.03)
	Sons	Of household head; discrete	3.91 (0.21)

a. For consistency, kin or community *ties* describe local relationships or connectivity, whereas kin or livelihood *networks* describe non-local relationships or connectivity.