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ORIGINAL PAPER

Critical evaluation of a long-term, locally-based wildlife monitoring program in West Africa

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Abstract Effective monitoring programs are required to understand and mitigate biodiversity declines, particularly in tropical ecosystems where conservation conflicts are severe yet ecological data are scarce. "Locally-based" monitoring has been advanced as an approach to improve biodiversity monitoring in developing countries, but the accuracy of data from many such programs has not been adequately assessed. I evaluated a long-term, patrol-based wildlife monitoring system in Mole National Park, Ghana, through comparison with camera trapping and an assessment of sampling error. I found that patrol observations underrepresented the park's mammal community, recording only two-thirds as many species as camera traps over a common sampling period (2006–2008). Agreement between methods was reasonable for larger, diurnal and social species (e.g., larger ungulates and primates), but camera traps were more effective at detecting smaller, solitary and nocturnal species (particularly carnivores). Data from patrols and cameras corresponded for some spatial patterns of management interest (e.g., community turnover, edge effect on abundance) but differed for others (e.g., richness, edge effect on diversity). Long-term patrol observations were influenced by uneven sampling effort and considerable variation in replicate counts. Despite potential benefits of locally-based monitoring, these results suggest that data from this and similar programs may be subject to biases that complicate interpretation of wildlife population and community dynamics. Careful attention to monitoring objectives, methodological design and robust analysis is required if locally-based approaches are to satisfy an aim of reliable biodiversity monitoring, and

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there is a need for greater international support in the creation and maintenance of local monitoring capacity.

Keywords West Africa · Camera trap · Detection bias · Locally-based monitoring · Law enforcement patrols · Mammal monitoring · Protected area management · Sampling error · Survey methods · Wildlife conservation

Introduction

Global declines in biodiversity are increasingly well documented and threaten the welfare and resilience of ecological and human communities (Balmford and Bond 2005; Butchart et al. 2010). Dependable monitoring programs are required to understand the extent and drivers of these declines, guide management action to slow or stop them, and assess the effectiveness of such conservation interventions. Despite international commitments to monitor and protect biodiversity, current ecological monitoring efforts are generally inadequate and biodiversity conservation targets are not being met (Lindenmayer and Likens 2009; Butchart et al. 2010). The establishment of protected areas has been society's chief response to biodiversity declines, yet the success of these interventions in adequately conserving species and ecosystems is increasingly questioned, and data necessary to evaluate their effectiveness are frequently lacking (Chape et al. 2005; Gaston et al. 2008). Similarly, efforts to integrate conservation with development and poverty reduction also often lack appropriate monitoring mechanisms for tracking their progress (Kremen et al. 1994; Wells and McShane 2004).

Recent increases in establishment of biodiversity monitoring programs have been accompanied by debate over their appropriate design. Proponents of a "strong inference" approach have emphasized the need for focused and experimental programs linking monitoring, management and research, whereby predictions from a priori hypotheses about causal relationships and underlying mechanisms are tested through system manipulations, and sources of uncertainty are carefully addressed (Yoccoz et al. 2001; Nichols and Williams 2006; Lindenmayer and Likens 2009). Other authors have argued that narrowly focused, manipulative monitoring programs are ill-suited to address the cumulative impacts of multiple anthropogenic stressors operating on diverse taxa across large spatial and temporal scales, nor are they likely to contend with unanticipated future changes to human and natural systems (Boutin et al. 2009; Haughland et al. 2010). Still others have raised pragmatic concerns about the ability of "professional" monitoring programs to effectively cover the spatial and temporal scales demanded by global gaps in biodiversity monitoring, and have instead promoted more participatory efforts (Danielsen et al. 2005; Schmeller et al. 2009).

Models of participatory or "locally-based" monitoring have been advanced as a means of addressing an apparent "conflict between scientific ideals and practical realities" in developing countries (Danielsen et al. 2003; Sheil and Lawrence 2004; Brashares and Sam 2005). Regions prone to conservation conflict—with high biodiversity and rapidly expanding human impacts—are disproportionately located in tropical and developing countries, where ecological data are typically scarce and monitoring programs most urgently needed (Balmford et al. 2001; Collen et al. 2008). These regions also frequently lack the institutions, funding, and technical capacity to implement the kind of professional scientific monitoring programs designed in wealthier countries (Barrett et al. 2001; Sheil 2001). In such areas, reliance on foreign professionals may be neither effective nor desirable, entailing unrealistically high implementation costs and low chances of sustainability, and failing to adequately engage or inform local resource users or managers who ultimately determine conservation outcomes. Locally-based schemes are defined by an emphasis on the participation of local stakeholders but can take many forms, including volunteer surveys, hunter reports, and traditional indigenous systems (Danielsen et al. 2005, 2009). While there are encouraging signs of the potential effectiveness of local monitoring programs in aiding management decisions, abating conservation threats, and empowering local communities to improve their livelihoods, a key outstanding question centers on their ability to deal with sampling error and thus reliably detect true trends in monitored populations (Yoccoz et al. 2003; Danielsen et al. 2005).

In this study, I evaluate the reliability of locally-based monitoring through a case study of an acclaimed patrol-based wildlife monitoring program in West Africa. Wildlife monitoring and research have received relatively little attention in West Africa, and the resulting scarcity of scientific information hinders management planning and response to the region's widespread hunting and human-wildlife conflicts (Ntiamoa-Baidu 1987; Oates 2002). An exception to the paucity of data comes from a long-term monitoring program in Ghana: the Wildlife Division of the Forestry Commission of Ghana (hereafter Ghana Wildlife Division or GWD) has been monitoring illegal hunting and mammal populations in protected areas under its jurisdiction for several decades (Asibey 1971; Brashares et al. 2001; Jachmann 2008a, b). The program is based on observations made during regular law-enforcement patrols, representing a form of "surveillance" monitoring (cf. Gray and Kalpers 2005; Keane et al. 2011), and has been described as a successful example of locally-based monitoring (Brashares and Sam 2005; Danielsen et al. 2005). Resulting data have been used to infer patterns and drivers of population decline and extinction, study trophic interactions, and assess management effectiveness (Brashares et al. 2001, 2004, 2010; Brashares 2003; Jachmann 2008a, b; Craigie et al. 2010; Burton et al. 2011). Nevertheless, the accuracy and precision of this monitoring system have not been formally evaluated.

Here, I present an assessment of the GWD monitoring program in Ghana's largest protected area, Mole National Park. Specifically, I compared results of recent data from the patrol-based system with those from a concurrent camera-trap survey, representing an alternative "professional" survey method that is increasingly being used to monitor mammal populations (O'Brien et al. 2010; O'Connell et al. 2010). I assessed concordance between methods with respect to estimates of mammal species richness, diversity, and relative abundance, as well as incidence of illegal hunting. I further compared spatial variation in estimated patterns as well as their relation to heterogeneity in species attributes, and I explored the potential influence of sampling error on trend estimation from the patrol monitoring data. The results provide insight into the strengths and weaknesses of Ghana's long-term mammal monitoring data, and have important implications for the design, implementation, and interpretation of locally-based wildlife monitoring.

Methods

Patrol monitoring data

The Ghana Wildlife Division's law enforcement monitoring system consists of observations of illegal activity and wildlife made by park staff during regular "anti-poaching" patrols within its wildlife protected areas (Brashares and Sam 2005; GWD 2005;

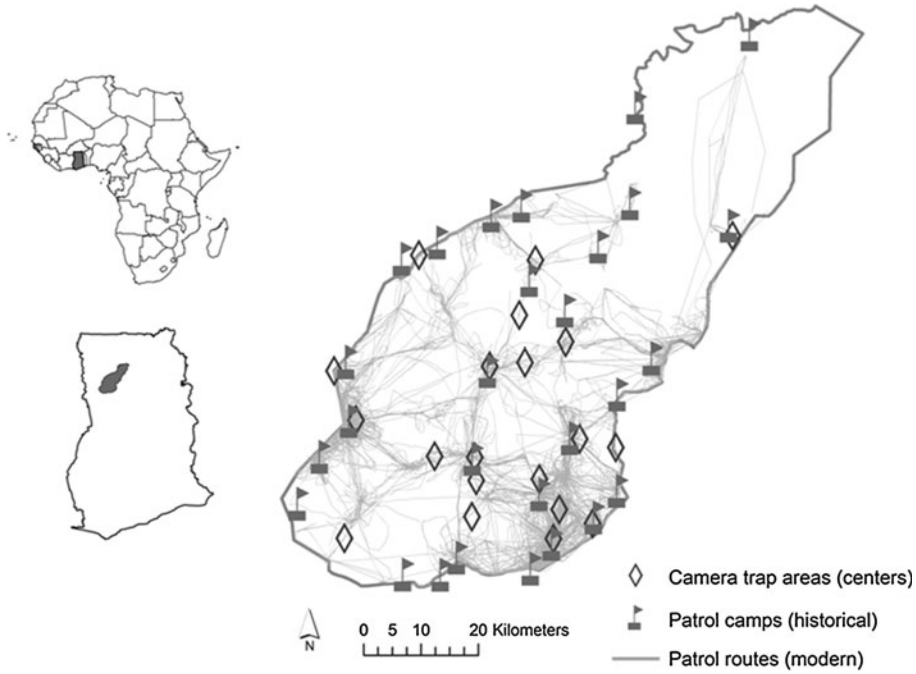

Fig. 1 Study area and sampling map, showing location of Ghana in Africa (*top left*), Mole National Park in northern Ghana (*bottom left*), and the distribution within the park of camera-trap sampling areas, approximate patrol routes for the period of methodological comparison (2006–2008), and patrol camps from which monthly summary reports were created for the historical dataset (1968–2001)

Jachmann 2008a, b). The system began in the late 1960s (Pegg 1969; Asibey 1971) and continues to the present day, although specific protocols of data collection across the entire period of monitoring are not well documented. The general scheme involves daytime foot patrols by teams of 3–5 "Wildlife Guards" that record sightings of mammal species and hunters (or hunting sign such as footprints, traps, etc.) while patrolling from camps distributed across the parks. A particular target group of monitored species has not been well defined but is generally conveyed as "larger" mammals (e.g., $\geq$1 kg, Brashares et al. 2001, 2004; "similar or larger size than a Maxwell's duiker *Cephalophus maxwelli*", Jachmann 2008a). While some amount of monitoring data exist for at least nine protected areas in Ghana (Brashares et al. 2001; Jachmann 2008b), I focused on data from Mole National Park (hereafter MNP), Ghana's largest protected area, encompassing ~4,600 km^2 of woodland savanna habitat in the country's Northern Region (Fig. 1).

MNP monitoring data were available for the period covering 1968–2008, but this study focused primarily on a subset of data from ~1,400 patrols (1,612 patrol-days) conducted between October 2006 and May 2008. These data provided the most overlap with the comparative camera trap survey (described below), and, unlike older patrol data, were spatially explicit (but generally representative of the full dataset; online Appendix 1). Patrol teams walked routes covering much of the park (Fig. 1), using handheld GPS units to record their positions at periodic intervals and at locations of observations of mammal species and illegal activities. Indices of relative abundance were calculated as the number of individuals counted (per species or pooled across species) divided by the number of patrols conducted as a measure of sampling effort (i.e., a catch-per-unit-effort or CPUE index; cf. Jachmann 2008a; Keane et al. 2011).

Camera trap survey

I compared data from MNP's patrol monitoring system with results of a camera trap survey conducted between October 2006 and January 2009. Data were obtained from 253 camera stations deployed in 32 spatially or temporally differentiated groups targeting different portions of the park and different seasons (Fig. 1). Within each group stations were set systematically at approximately 1-km intervals and were active for a mean of 21.6 days (SD = 12.8), yielding a total survey effort of 5,469 trap-days. Sampling effort was highest in central and southeastern portions of the park and during dry season months of November to March, which corresponded with the spatial and temporal intensity of effort in the patrol monitoring system (Fig. 1). Indices of relative abundance were calculated as the number of individuals photographed divided by the number of camera-trap days (analogous to the patrol CPUE index). Further details on the camera sampling are described in Burton et al. (2011).

Comparative analysis

I first tabulated the number of mammal species detected by each method and compared these estimates of species richness. I then compared indices of relative abundance across species for both methods and combined richness and abundance by calculating two common measures of species diversity (Simpson and Shannon-Wiener; Krebs 1999).

Given that the period of comparison was too brief to allow assessment of temporal trends, I substituted space for time and compared spatial patterns in richness, relative abundance, and turnover (dissimilarity) across the park as discerned by the two methods. MNP had previously been spatially subdivided for management purposes into 24 sectors of roughly similar size (mean size = 188 km^2; GWD 2005). I used these sectors as management-relevant sampling units for spatial comparison, pooling data from all camera stations or patrol observations falling within a given sector. To create a spatially explicit measure of patrol effort for the CPUE index, I re-created patrol routes from corresponding GPS locations, divided routes into equal 200 m segments, and summed the number of patrol segments within a given sector.

I compared estimates of species accumulation and turnover across management sectors between cameras and patrols, as well as relationships between distance to park edge (from sector centroids) and mammal diversity and relative abundance (summed across all species). Such measures relate to inference likely to be drawn from a monitoring program (e.g., changes in community structure, anthropogenic edge effects), and thus their comparison provides insight into the reliability of such inference (cf. Kremen et al. 2011). The R package *vegan* was used to calculate accumulation curves and Bray-Curtis dissimilarity matrices, and to apply a Mantel test to assess correlation between patterns of community dissimilarity across sectors (based on 999 matrix permutations; Oksanen et al. 2011).

I evaluated the comparative measures in relation to three species traits expected to affect detectability: body mass, daily activity pattern (diurnal vs. nocturnal or crepuscular), and social group size (Table S1 in supplementary material). Trait data were obtained from the PanTHERIA database (Jones et al. 2009; supplemented by species-specific sources where necessary, see Burton et al. 2011). Given that abundance indices and trait data were not normally distributed, I used non-parametric statistics to assess correlations and compare means (Spearman rank correlation and Wilcoxon rank-sum test, respectively). Statistical analyses were performed in program R version 2.14.0 (R Core Development Team 2011), and analyses of spatial GIS data were done in ArcGIS version 9.3.1 (ESRI, USA).

Effect of sampling variation on patrol monitoring trends

In addition to the methodological comparison between modern patrol and camera-trap data, I assessed the potential effect of sampling variation on interpretation of long-term trends from the monitoring data. Trend estimation to assess population viability or the importance of environmental drivers is a common objective of monitoring programs, but one often made difficult by the confounding of variation due to environmental and population processes with that caused by sampling error (Clark and Bjørnstad 2004). Variable observer effort is a common source of sampling bias affecting the detection probability of individuals in a sampled population, and it may be particularly important for monitoring protocols with relatively low levels of standardization, such as GWD's patrol-based system (Keane et al. 2011). Previous analyses of GWD monitoring data have used different approaches to deal with variation in sampling effort, from relying on an assumption of constant effort across space and time (e.g., Brashares et al. 2001, 2004) to applying a strict standardization of effective man-hours on patrol (Jachmann 2008a, b). I explored the effect of accounting for sampling effort by comparing trends derived from uncorrected counts with those from the CPUE index of count per patrol.

Even with standardized effort, other aspects of sampling can introduce heterogeneity in detection probabilities and thereby influence the relationship between a count and true abundance (Pollock et al. 2002). One approach to dealing with detection heterogeneity is to use repeated sampling of a site over a short enough period that it can reasonably be considered "closed" to changes in population status. Differences between replicate samples are then assumed to represent sampling error around the true but unknown number of individuals at the site (Morris and Doak 2002; Kéry et al. 2009). While such replicate sampling was not an explicit part of the GWD monitoring program design, I applied this approach in a post hoc assessment of detectability for a sample of MNP monitoring data.

I examined a random subset of data for four species with different expected detectabilities (following Brashares and Sam 2005): olive baboon (*Papio anubis*) and African buffalo (*Syncerus caffer*) are relatively large, abundant and conspicuous species, whereas oribi (*Ourebia ourebi*) is a small and secretive antelope of intermediate abundance, and leopard (*Panthera pardus*) is a rare and secretive felid with notoriously low detectability. The level of spatial and temporal resolution in the patrol dataset was coarser for the period 1968–2001 (monthly observation summaries for individual patrol camps; online Appendix 1), so I conducted separate analyses on these "historical" data and the finer-scaled "modern" data collected from 2004 to 2008 (consisting of observations along individual patrol routes with GPS locations, as described above). For historical data, I made the simplifying assumption that counts from one patrol camp in consecutive months within a common season (defining "wet" as June–August and "dry" as December-February to avoid transitional months) should be sampling the same subpopulations (i.e., no migration for that area over that time period). I randomly selected a set of 20 such "replicate" counts for each of the four focal species (from different camps and seasons, with approximately equal patrol effort between matched pairs) and calculated the difference between paired replicate counts as a crude estimate of sampling error. For modern data (2004–2008), I estimated the coefficient of variation for the four focal species from a random sample of counts (n = 11–25) from different patrol days within the same month and management sector (i.e., considered to be replicates).

Results

Species richness, diversity and abundance

Observations of 20 mammal species were recorded by the patrol monitoring system during the period of methodological comparison (Oct. 2006 to May 2008; Fig. 2; Table S1 in supplementary material). By contrast, a third more mammal species were detected during the camera trap survey. Three species infrequently recorded by the patrol system were not detected by cameras and 13 photographed species were not included in the patrol data, thus there were 17 species detected by both methods, and both also detected illegal hunting activity (Table S1 in supplementary material). Richness estimates were similar for ungulates and primates but differed considerably for carnivores and rodents (Fig. 2; Table S1 in supplementary material). Mean body mass across detected species differed significantly between the two methods, with camera traps detecting more smaller-bodied species than patrols (one-sided Wilcoxon rank-sum test, $W = 199.5$, $P = 0.024$; cameras: median = 12.8 kg, range = 0.22–3,825 kg, $n = 30$; patrols: median = 47.8 kg, range = 3.7–3,825 kg, $n = 20$; Fig. 3). Species' daily activity patterns and group sizes also contributed to differences in detectability between methods. Camera traps detected 11 nocturnal species (of 28 for which activity pattern descriptions were available; Table S1 in supplementary material) while patrols only detected 3 (of 20; one-sided binomial proportions test $\chi^2 = 2.26$, $P = 0.066$; Fig. 3), and average group size was larger across species detected by patrols than for those detected by cameras (patrols: median = 10.4; cameras: median = 2.0; one-sided Wilcoxon rank-sum test, $W = 199.5$, $P = 0.062$; Fig. 3).

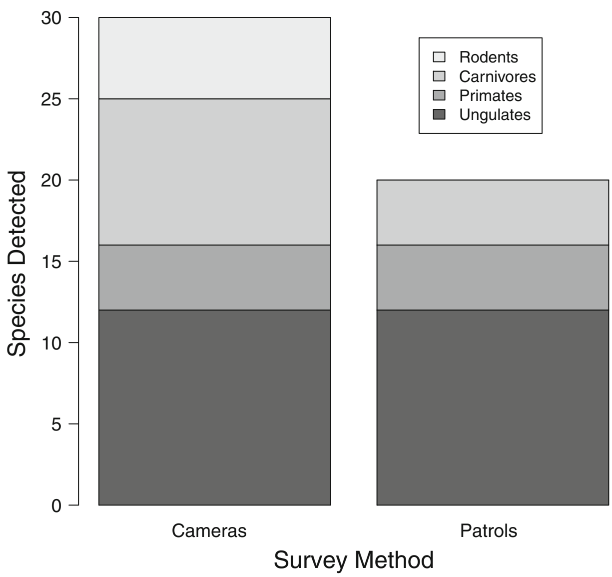

Fig. 2 Comparison of species richness estimates for Mole National Park from the camera trap survey and patrol monitoring observations (for the 2006–2008 period of comparison). For simplicity elephant and aardvark are included under "Ungulates" and scrub hare under "Rodents"

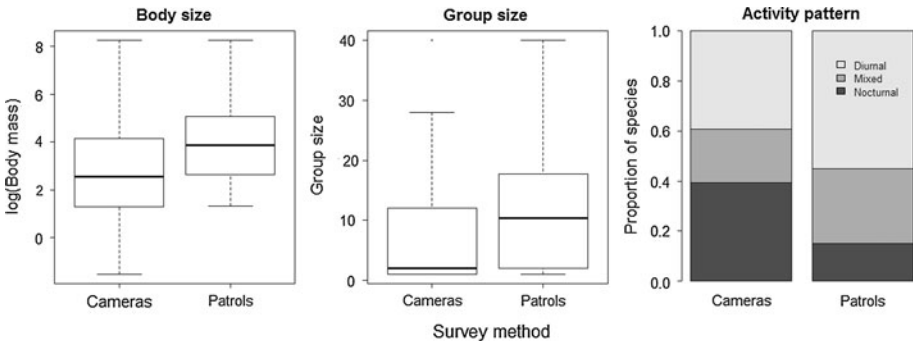

Fig. 3 Summary of trait values across species detected in Mole National Park by the camera trap survey and patrol monitoring observations (for the 2006–2008 comparison period). *Boxplots* show the median value, interquartile range, and full range of data

Estimates of relative abundance varied widely across species and between the two methods, with methodological discrepancies related to species traits as seen for richness estimates (Fig. 4). There was a significant positive correlation between abundance indices from the two methods (Spearman $r_s = 0.55$, $P < 0.001$ when all 33 species and humans were included; $r_s = 0.58$, $P = 0.014$ when including only the 17 species in common). However, there was considerable scatter in the relationship, with strong agreement for some species (e.g., kob, *Kobus kob*, and olive baboon) and large disparities for others (e.g., spotted hyena, *Crocuta crocuta*, and bushbuck, *Tragelaphus scriptus*; Fig. 4; Table S1 in supplementary material). Measures of species diversity combining richness and relative abundance were higher for the camera trap survey

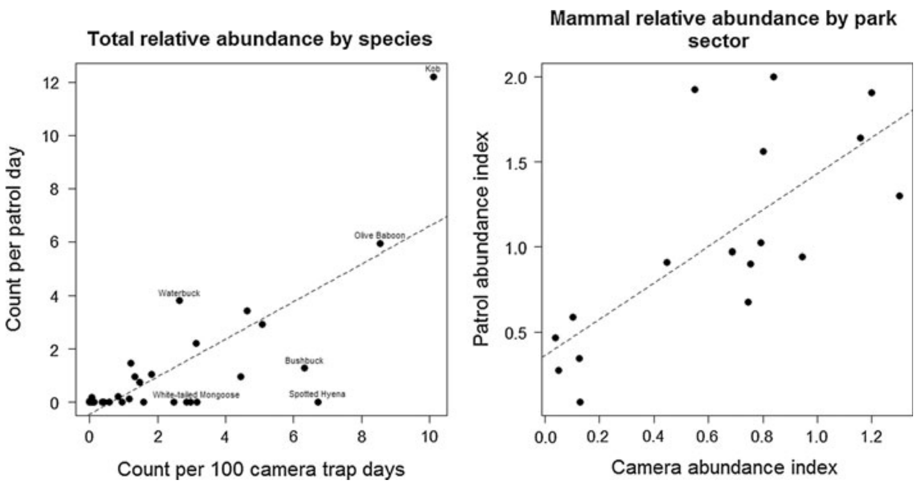

Fig. 4 Relative abundance indices generated from patrol and camera-trap data in Mole National Park (over 2006–2008 period of comparison). *Left panel* shows total abundances (per unit effort) across all 34 species detected (including humans, with select species highlighted to illustrate strong or weak concordance). *Right panel* shows abundance indices across all mammal species for 17 park management sectors within which both methods detected at least one species. *Dashed lines* give linear fits ($R^2 = 0.58$ for *left panel*, $R^2 = 0.54$ for *right panel*)

(Shannon-Wiener: $H_{cameras} = 2.91$, $H_{patrols} = 2.17$; Simpson's reciprocal index: $1/D_{cameras} = 14.6$, $1/D_{patrols} = 6.1$).

Spatial patterns

Indices of total mammal abundance for patrol and camera data were significantly correlated across the 17 management sectors sampled by both methods ($r_s = 0.75$, $P < 0.001$; Fig. 4). Community dissimilarity (or turnover) across sectors was also correlated between methods (Mantel $r = 0.60$, $P = 0.001$). Conversely, indices of species richness and diversity were spatially uncorrelated between patrols and cameras (species per unit effort, $r_s = -0.005$, $P = 0.99$; Simpson's $1/D$, $r_s = -0.14$, $P = 0.59$), with the latter result driven by the relative dominance of certain common species in the patrol data (e.g., kob, baboon; Table S1 in supplementary material). Both methods indicated an increase in mammal abundance with increasing distance from the park boundary (i.e., negative edge effect; patrols $r_s = 0.49$, $P = 0.048$; cameras $r_s = 0.52$, $P = 0.031$), whereas only camera data showed an edge effect on mammal diversity (correlation between Simpson's $1/D$ and distance from edge; patrols $r_s = -0.39$, $P = 0.12$; cameras $r_s = 0.49$, $P = 0.046$).

Across-sector correlations in patrol- and camera-derived abundance indices were generally much stronger for larger-bodied, social, and diurnal species (such as many of the larger ungulates) than for smaller, nocturnal and solitary species (like carnivores; Table S1 in supplementary material). Species accumulation curves (calculated across sectors) leveled off for both methods at their respective estimates of total richness (Fig. S1 in supplementary material), implying that they were unlikely to detect more species. Correspondence between indices of hunting activity across sectors was very weak ($r_s = -0.16$), with cameras only sporadically detecting hunters.

Effect of sampling variation

There was considerable temporal variation in patrol effort in the long-term monitoring dataset, with a general trend of declining effort over time ($r_s = -0.38$, $P = 0.026$), but more specifically a period of particularly low effort in the mid-1980s (Fig. 5). Correcting for effort had a significant impact on the interpretation of temporal trends, as total annual counts pooled across 33 mammal species (Table S1 in supplementary material) largely followed variation in patrol effort, declining strongly in the mid-1980s before increasing more recently (Fig. 5). Accounting for sampling effort (by dividing by the annual number of patrols) changed the pattern from a generally declining trend ($r_s = -0.17$) to one that tended toward an increase ($r_s = 0.25$, though neither correlation was statistically significant, $P > 0.15$; Fig. 5). A confounding influence of sampling effort was also seen in the trend of illegal hunting activity: while uncorrected annual counts of hunting sign and those corrected for patrol effort both showed a general increasing trend over time, the increase in the latter was much stronger ($r_s = 0.30$, $P = 0.083$ for uncorrected counts, $r_s = 0.50$, $P = 0.0027$ for the CPUE index). In addition to temporal sampling variation, there was considerable spatial variation in recorded effort levels across the 27 patrol camps from which historical data were available (range = 19–2,841 patrols per camp, median = 857), indicating that correcting for effort before interpreting spatial patterns over time would also be very important.

My assessment of potential sampling error due to detectability suggested considerable variation in consecutive patrol counts within a sampling unit (assumed to be replicate samples of the same local sub-populations). Differences between paired replicate counts

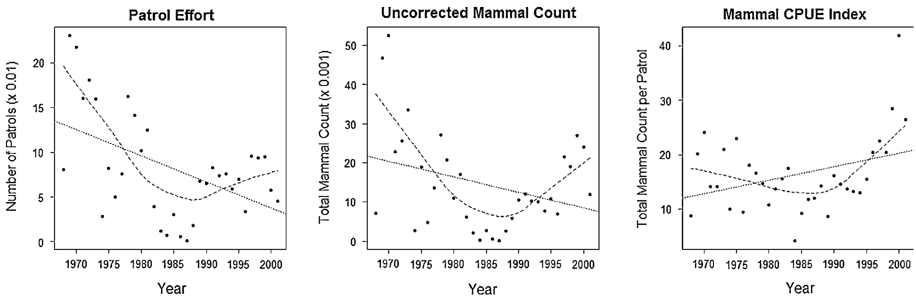

Fig. 5 Trends in annual patrol effort (*left*), total mammal counts (*middle*), and counts corrected for effort (CPUE index, *right*) across 33 mammal species (pooled counts) in Mole National Park over the historical monitoring period (1968–2001). Differing interpretations of overall trend in mammal abundance are highlighted by linear fits (*dotted lines*), while the effect of a decline in patrol effort during the middle of the monitoring period is emphasized in the smoothed lowess fits (*dashed lines*)

randomly sampled from the historical dataset were substantial for all four species examined, representing on average around one half or more of the magnitude of the means of paired counts [difference between pairs was an average of 56 % of the pair mean for baboons (range 7–153 %), 79 % for buffalo (range 3–198 %), 67 % for oribi (range 0–124 %), and 37 % for leopard (0–133 %), with only 10 paired monthly counts available for leopard; Table S2 in supplementary material]. Counts were similarly variable in the modern replicate samples (2004–2008), with large coefficients of variation for baboon (49), buffalo (74), and oribi (54; Table S2 in supplementary material). The modern dataset contained only 17 records of leopard, of which only 7 were sightings and few were close in space or time, confirming that detectability was likely very low for this species (see also Burton et al. 2011).

Discussion

Reliability of patrol-based wildlife monitoring data

These results sound a cautionary note for use of wildlife data generated from MNP's law enforcement monitoring program, and for similar locally-based monitoring efforts elsewhere. While reinforcing previous assertions that the program generates a large amount of otherwise unavailable information for this regionally important protected area (e.g., Brashares and Sam 2005), I found the data subject to biases that warrant careful analysis and interpretation. MNP patrol observations tended to systematically underestimate the presence and abundance of important members of the medium- and large-bodied mammalian community, such as most predators, and even counts of well-detected species were affected by substantial sampling variation in space and time. This sampling error may obscure underlying ecological patterns, inflating uncertainty around observed population or community dynamics, and thereby potentially limiting the usefulness of monitoring data for addressing management concerns. Patrol monitoring and camera trapping showed some correspondence in observed patterns of management interest (e.g., community turnover, edge effect on total abundance), but other significant discrepancies were apparent (e.g., total richness, edge effect on diversity). This is consistent with several other comparative studies of locally-based or participatory monitoring approaches that have noted methodological limitations and a need for cautious interpretation (Noss et al. 2004; Can and Togan 2009; Mueller et al. 2010; Crall et al. 2011; Kremen et al. 2011). Given that detection bias and large sampling variance are likely common problems for many wildlife monitoring programs, particularly those facing challenging socioeconomic constraints, more attention to methodological rigor and monitoring effectiveness seems warranted (Yoccoz et al. 2001, 2003; Pollock et al. 2002; Keane et al. 2011).

Program strengths and study limitations

The primary aim of GWD's patrol system is deterrence of illegal hunting within protected areas (i.e., law enforcement), and the results of this study suggest it is effective at detecting signs of hunting activity (at least relative to camera trap sampling, although more assessment is warranted, e.g., Gavin et al. 2010). The system also seemed to perform relatively well at monitoring large, diurnal mammals (such as many ungulates and primates), which represent high-value targets of bushmeat hunting and a traditional focus of wildlife management (Jachmann 2008a, b). It may not be surprising that patrols did not reliably detect more elusive species like carnivores, and other comparative studies have noted that detectability—and hence correspondence between different methods—tends to increase with body size (e.g., Silveira et al. 2003). It is important to recognize that, like all survey methods, camera trapping sampled the park's populations with error, and resulting data may not be the best representation of "truth" against which to compare the patrol data—the question of what constitutes an acceptable level of agreement between different methods warrants further attention. Moreover, my analyses were relatively simple and more detailed work is needed to assess consequences of the observed sampling error for interpretation of trends from the patrol monitoring data over different temporal and spatial scales.

Consideration of other benefits of locally-based monitoring

Irrespective of the GWD monitoring program's current capability to account for sources of sampling error and accurately track wildlife populations, the program produces other important benefits. These include basic tangibles like providing employment in a region with considerable poverty, as well as specific management tools such as an ability to track staff performance and increase motivation (Jachmann 2008b). In fact, proponents of locally-based monitoring systems stress that a predominant emphasis on the generation of robust wildlife data is unrealistic and inappropriate for many monitoring programs in developing nations (Sheil 2001; Danielsen et al. 2003, 2005). They highlight other important (and interrelated) features of successful locally-based programs, such as long-term sustainability, cost-effectiveness, involvement of local stakeholders, and ease of incorporation into management decisions. However, a critical assessment of these in relation to the GWD monitoring system reveals significant shortcomings, such as susceptibility to changing donor priorities, gaps in basic operational funding, conflicts with local communities, and uncertain links to management action (online Appendix 2). The GWD program may thus not represent an ideal model of effective locally-based monitoring, despite its noted strengths. In fact, a simple dichotomy between locally-based and professional scientific monitoring systems may not be useful in practice since many programs combine elements of both and should be assessed based on their distinct features and the specific context in which they operate (Danielsen et al. 2009).

Recommendations for more effective monitoring

The formation of explicit objectives and use of methodologies capable of meeting them are critical components of effective monitoring (Yoccoz et al. 2001). It may be more tractable for a program like GWD's to focus on carefully chosen "indicator" species linked to particular management questions, rather than on the entire larger mammal community (cf. Gray and Kalpers 2005), although narrow focus on a few "game" species may not be appropriate given the significance of diverse taxa to both ecosystem functioning and local livelihoods (Asibey 1974; Sinclair and Byrom 2006). Regardless of target species, incorporating an assessment of sampling error into the protocol would improve inference. A small proportion of patrol routes could be explicitly designed as replicate samples to monitor sampling variation, and techniques such as distance sampling could be used to estimate detection probability (Buckland et al. 2001). Covariates known to affect detectability could be monitored (analogous to current tracking of patrol effort; Jachmann 2008a, b) and used in analytical approaches that model observation processes distinctly from underlying ecological dynamics (e.g., Kéry et al. 2009). Limiting sampling variation by keeping effort more consistent over time and space would also be helpful. While simplicity is key to successful patrol-based monitoring, linking management to local technical support and building capacity in requisite skills (such as through local university wildlife programs) could be of great benefit to monitoring effectiveness.

Inference from monitoring data is also strengthened by linking a program to specific management questions defined by conceptual models and a priori hypotheses (Nichols and Williams 2006; Lindenmayer and Likens 2009). A recent MNP management plan briefly promotes the concept of adaptive management (GWD 2005, p. 87) but does not identify particular questions or models to be tested by the monitoring data, even though such models are implicit in the plan (e.g., effect on mammal populations of hunting pressure, water availability, and fire frequency). Specific monitoring of key predictor variables, including both environmental and socioeconomic factors (Bawa and Menon 1997), would also help improve understanding of the greater park ecosystem, as would the facilitation of focused research programs to complement monitoring (i.e., test hypotheses identified by surveillance using a more experimental approach). Post hoc inference from unfocused surveillance monitoring has a poor ability to identify causal relationships (Nichols and Williams 2006) and may often be disassociated from particular program features if they are poorly documented (e.g., monitoring objectives, methodological protocols, sampling errors). In general, greater emphasis on making monitoring data more transparent and readily available (including metadata and means of error propagation) to both local stakeholders and the broader conservation community would be a great benefit.

Another potential improvement to locally-based monitoring programs like GWD's would be periodic use of complementary methods for testing or "calibrating" the relationship between patrol observations and more robust measurements of species abundance or richness (Pollock et al. 2002; Keane et al. 2011). This study demonstrates the usefulness of camera trapping, particularly for monitoring elusive species (i.e., smaller or nocturnal species, carnivores). Camera trapping provides additional benefits, including photographic evidence of animal occurrence that can be archived, assessed by outside experts, and used to increase awareness and enthusiasm for monitored wildlife. Resulting data can also be readily used in other studies, for example on animal morphology or activity patterns, and are well-suited to robust analytical frameworks like mark-recapture and occupancy modeling (O'Connell et al. 2010). On the other hand, camera trapping requires large initial investments in equipment and training and can be subject to various challenges, including technical malfunctions or inconsistent performance, theft or animal damage, and detection biases (Swann et al. 2004; Larrucea et al. 2007). Many other methods have been used to monitor mammals (e.g., Stoner et al. 2007; Long et al. 2008; Hoppe-Dominik et al. 2011), and studies comparing the effectiveness of different methods are of considerable use in guiding monitoring programs (Jachmann 2002; Barea-Azcón et al. 2007; Kindberg et al. 2009). While further methodological comparison is warranted, and choice of methods should be dictated by monitoring objectives, I suggest that camera trapping is a useful and reliable method that meets high standards of evidence (Burton et al. 2011).

Conclusion

While this study represents a relatively simple evaluation of one particular monitoring program, it has broad implications for wildlife monitoring efforts elsewhere. The GWD patrol system is implemented in many other parks in Ghana (Jachmann 2008b), and similar patrol-based programs are common elsewhere (Keane et al. 2011). Likewise, participatory community-based monitoring programs are increasingly being initiated or proposed in response to wildlife policy changes in Ghana (e.g., Sheppard et al. 2010) and are widespread and increasing across Africa and other parts of the world (Danielsen et al. 2005, 2009). Such efforts are sorely needed to inform conservation initiatives and promote sustainable livelihoods, but are likely to face similar challenges to the MNP monitoring, particularly in terms of limited funding and technical capacity, low animal detectabilities, and restricted resources to support data management and analysis. If these programs are to succeed in adequately tracking wildlife and other biological resources, it is a critical time for concerted focus and international support in the pursuit of reliable and practical methods for adaptive monitoring and management (Danielsen et al. 2009; Lindenmayer and Likens 2009).

My aim in this work is to recognize the importance of monitoring components of biological diversity, such as Ghana's larger mammal communities, and to stress the need for locally relevant and reliable monitoring programs in species-rich regions that have thus far received inadequate attention. It is necessary to capitalize on and strengthen existing efforts, like the GWD's mammal monitoring program, while also supporting creative new initiatives that increase local participation and relevance. However, it is equally important to emphasize that programs charged with the responsibility of informing society about the status of biodiversity must be capable of producing reliable inference. As noted by Nichols and Williams (2006), the identification of monitoring objectives and management actions should be based on the value judgments of a community of relevant stakeholders, but the remaining components of effective monitoring for conservation are largely the purview of ecological scientists and technical experts, who must receive adequate training and support, and work closely with stakeholders to provide the reliable information they need.

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Supplementary Information for "Critical evaluation of a long-term, locally-based wildlife monitoring program in West Africa", ***Biodiversity and Conservation***

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Contents:

Table S1. Mammal species detected in Mole National Park, Ghana, by the patrol monitoring system and camera trap survey, with species traits that affected detectability.

Common name	Scientific name[a]	Body mass (kg)[b]	Activity period [b]	Group size [b]	Relative abundance (count per unit effort)[c]			
					Cameras	Patrols (2006-08)	Patrols (1968-2001)	Spatial correlation[d]
Kob	*Kobus kob*	80.0	diurnal	40	10.11	12.21	1.50	0.60
Olive baboon	*Papio anubis*	17.7	diurnal	40	8.54	5.95	4.21	0.45
Spotted hyena	*Crocuta crocuta*	63.4	nocturnal	8	6.71	0.02	0.0003	-0.08
Bushbuck	*Tragelaphus scriptus*	43.3	mixed	1	6.31	1.28	0.67	-0.01
Warthog	*Phacochoerus africanus*	82.5	diurnal	5	5.08	2.93	1.29	0.84
Hartebeest	*Alcelaphus buselaphus*	160.9	diurnal	20	4.63	3.44	2.22	0.55
Elephant	*Loxodonta africana*	3824.5	diurnal	19.5	4.44	0.95	0.24	0.51
White-tailed mongoose	*Ichneumia albicauda*	3.6	nocturnal	1	3.14	0	0 [e]	-
Green monkey	*Chlorocebus sabaeus*	3.7	diurnal	12	3.13	2.22	1.21	0.56
Leopard	*Panthera pardus*	52.4	mixed	1	2.96	0.01	0.004	0.10

Large-spotted genet	*Genetta pardina*	2.0	nocturnal	1	2.85	0	0.003[f]	-
Waterbuck	*Kobus ellipsiprymnus*	204.4	mixed	12	2.63	3.82	1.58	0.81
Crested porcupine	*Hystrix cristata*	13.4	nocturnal	1	2.47	0	0.001[f]	-
African buffalo	*Syncerus caffer*	592.7	mixed	12	1.81	1.05	1.48	0.28
Aardvark	*Orycteropus afer*	56.2	nocturnal	1	1.57	0	0.0003	-
Human [g]	*Homo sapiens*	-	-	-	1.46	0.76	0.03	-0.16
Roan antelope	*Hippotragus equinus*	264.2	diurnal	12	1.33	0.97	0.77	0.44
Patas monkey	*Erythrocebus patas*	8.0	diurnal	28	1.21	1.47	0.76	0.11
Red-flanked duiker	*Cephalophus rufilatus*	12.1	diurnal	1	1.17	0.13	0.30	0.13
Scrub hare	*Lepus saxatilis*	2.6	nocturnal	-	0.95	0	< 0.0001[f]	-
Grey duiker	*Sylvicapra grimmia*	15.6	mixed	1	0.82	0.22	0.29	0.15
African civet	*Civettictis civetta*	12.1	nocturnal	1	0.59	0	0.001	-
Marsh cane rat (grasscutter)	*Thryonomys swinderianus*	3.8	nocturnal	1	0.42	0	0.002	-
Marsh mongoose	*Atilax paludinosus*	3.6	mixed	1	0.37	0	0.03	-
Caracal	*Caracal caracal*	12.0	nocturnal	1	0.37	0	< 0.0001	-

Gambian mongoose	*Mungos gambianus*	1.6	diurnal	6.7	0.13	0	0[e]	-
Striped ground squirrel	*Xerus erythropus*	0.6	-	-	0.11	0	0	-
Side-striped jackal	*Canis adustus*	10.4	nocturnal	2	0.07	0.004	< 0.0001	0.49
Oribi	*Ourebia ourebi*	17.2	diurnal	2	0.05	0.17	0.78	0.01
Senegal galago	*Galago senegalensis*	0.2	nocturnal	3.5	0.05	0	< 0.0001[f]	-
Giant pouched rat	*Cricetomys gambianus*	1.3	-	-	0.04	0	0	-
Geoffroy's black and white colobus	*Colobus vellerosus*	7.7	diurnal	16	0	0.04	0.13	-
Lion	*Panthera leo*	158.6	nocturnal	8.7	0	0.01	0.01	-
Bohor reedbuck	*Redunca redunca*	43.3	mixed	4	0	0.0006	0.04	-
Red river hog	*Potamochoerus porcus*	70.0	mixed	10.6	0	0	0.06[f]	-
Hippopotamus	*Hippopotamus amphibius*	1536.3	mixed	-	0	0	0.004	-
Yellow-backed duiker	*Cephalophus silvicultor*	62.0	mixed	1	0.00	0.00	0.002	-
Wild dog	*Lycaon pictus*	22.0	nocturnal	9.3	0.00	0.00	0.0007	-

"Long nose mongoose"[e]	-	-	-	-	-	-	0.02[e]	-
"Dwarf mongoose"[e]	-	-	-	-	-	-	0.001[e]	-

[a] Species nomenclature follows IUCN Red List 2010, Wilson & Reeder (2005). See also Grubb (1998).

[b] Average species trait values from Jones et al. (2009) when available (see also Burton et al. 2011). For activity period, "mixed" includes crepuscular.

[c] Indices of relative abundance calculated as number of individuals counted per unit of effort, which was set at 100 camera-trap days for the camera survey (5,469 total effort) and 1 patrol-day for the patrol counts (1612 total for the modern dataset and 28,225 for the historical).

[d] Spearman rank correlation coefficients for relative abundance estimates across management sectors from cameras and 2006-08 patrols.

[e] Mongoose species were not clearly or correctly identified in the historical patrol data since long nose and dwarf mongoose do not occur in Ghana (see Burton et al. 2011).

[f] Large-spotted genet was assumed for records of "genet" in the historical dataset, crested porcupine for records of "porcupine", Senegal galago for 1 record of "bushbaby", and scrub hare for 1 record of "rabbit". "Bush pig" was also assumed to be the same as "red river hog".

[g] Observations of humans included signs of illegal hunting activity for the patrol data (e.g., footprints, poaching camps, traps).

Table S2. Variation in consecutive counts considered to be randomly selected replicate samples of relative abundance for four species monitored by the Mole National Park patrol monitoring system that vary in expected detectability.

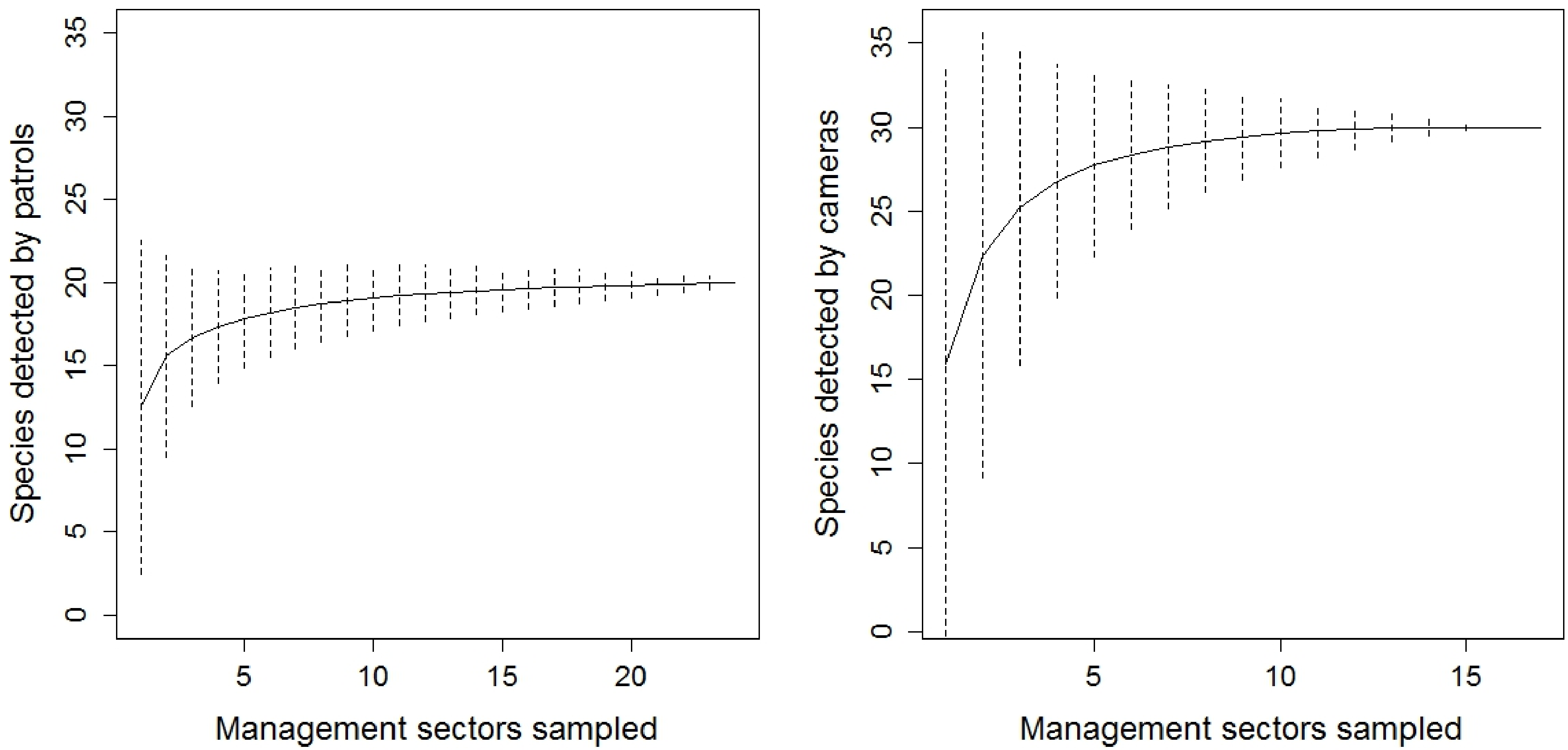
Species[a]	Historical patrols (1968-2001)[b]		Modern patrols (2004-08)[c]	
	Mean count (SD)	Mean difference between paired replicates (SD)	Consecutive counts	Coefficient of variation
Olive baboon	81.6 (81.8)	36.7 (49.6)	25	49
African buffalo	50.1 (64.8)	35.3 (45.3)	11	74
Oribi	15.5 (26.2)	7.1 (5.4)	14	54
Leopard	1.5 (1.0)	0.8 (1.3)	-	-

[a] Scientific names given in Table S1.

[b] Historical counts were a random sample of 20 paired counts for each species (10 for leopard), with each pair considered replicate samples of the same local subpopulation (i.e., same patrol camp, consecutive months; see Methods in main text).

[c] Modern counts were a random sample of consecutive counts made on different days within the same month and management sector for each species. There were insufficient detections of leopard for this assessment.

Figure S1. Accumulation curves for the number of species detected by patrols (left) and camera traps (right) across surveyed management sectors in Mole National Park (dashed lines represent 95% confidence intervals).



Appendix 1: Representativeness of modern patrol monitoring data

Data from the Ghana Wildlife Division’s patrol monitoring program in Mole National Park (MNP) were available for the period covering 1968-2008, but this study focused primarily on a subset of data from ~1,400 patrols (1,612 patrol-days) conducted between October 2006 and May 2008. To ensure the modern subset was generally representative of the long-term monitoring dataset, I compared estimates of species richness and relative abundance from the 2006-08 period with those from the entire 1968-2008 monitoring period (~31,000 patrols).

The subset of patrol data used in the methodological comparison with camera trapping were generally representative of the longer-term MNP patrol monitoring system as a whole. The historical dataset (1968-2001) included observations of 35 mammal species (Table S1); however, of the 15 species that did not occur in the 2006-08 subset, 9 had less than 10 total observations (including 4 with only 1 record) and only 2 had more than 50 observations over the 33-year period (red river hog, *Potamochoerus porcus*, which likely no longer occurs in the park; and hippopotamus, *Hippopotamus amphibius*, with only a few individuals potentially occurring in the Kulpawn river at the northern edge of the park; GWD 2005). Furthermore, few of the 15 species missing from the modern data had recent records in the historical dataset, and there is some taxonomic uncertainty associated with several of them (e.g., mongooses; see Table S1 notes). The 15 most abundant species were the same for both the historical and modern datasets, and species’ rank abundances were very similar (Table S1). Further details on the MNP monitoring dataset are given in Burton (2010).

Appendix 2: Consideration of other potential benefits of locally-based monitoring for the Ghana Wildlife Division's patrol monitoring system.

Proponents of locally-based monitoring systems stress that a predominant emphasis on the generation of robust wildlife data is unrealistic and inappropriate for many monitoring programs in developing nations (Danielsen et al. 2003, 2005). As part of my assessment, I briefly considered other potential benefits of locally-based monitoring for the case of Ghana Wildlife Division's (GWD) monitoring system. Specifically, I considered themes of long-term sustainability, cost-effectiveness, involvement of local stakeholders, and ease of incorporation into management decisions.

With regard to sustainability, persistence of the GWD program for over four decades in a developing region where wildlife management has not generally been a priority is quite remarkable. Nevertheless, further scrutiny raises warning flags about the program's performance over time. For instance, the historical dataset shows great variation in the number of monthly reports filed over time, with a period of particularly poor reporting in the 1980s. While it is possible that this reflects loss or displacement of completed reports rather than true variation in monitoring effort, the period corresponds to a time of broader economic decline in Ghana when "the park experienced considerable problems […] infrastructure was neglected and poaching was virtually uncontrolled" (GWD 2005: 18). Other sources also point toward poor management capacity and low staff morale at that time (e.g., Jamieson 1987). Periods with greater recorded effort correspond to the initial momentum of the program (Pegg 1969; Asibey 1971) and more recent support from international donor-funded projects (GWD 2005). These correlations signal that, unsurprisingly, the program's effectiveness vacillates with the broader economic context within which park management is situated. This highlights the fact that locally-based monitoring programs are not immune to the effects of international economic influences like changing donor priorities, and that securing sustainable sources of funding remains a common priority (Rodriguez 2003; Mortensen and Jensen 2012).

On the topic of funding, the low cost of many locally-based programs has been highlighted as a key feature (Danielsen et al. 2003, 2005). I have not conducted a detailed cost assessment for the Mole National Park (MNP) monitoring program, but Brashares and Sam (2005) and Jachmann (2008a) indicate a relatively low cost on the order of US$1-15/km^2/year. While this would appear to be a positive feature of the program, it is worth asking if this level of funding is sufficient. Notwithstanding the recommendation to better address sampling error (and the associated argument that data quality influences the cost-effectiveness of monitoring; Nichols and Williams 2006), there are signs that more operational funding is required. For example, there are program needs in areas such as basic equipment for field staff (e.g., hiking boots, camping gear), transportation costs (e.g., fuel, vehicle repairs), and capacity for effective data management (e.g., decaying historical reports, incomplete databases; Burton 2010). Such problems are not unique to the GWD program and I use them only to highlight the widespread need for greater financial and technical support of tropical biodiversity monitoring efforts (Balmford et al. 2003; Balmford and Whitten 2003). As noted above, the impact of international donor-funded projects can be substantial yet often unsustainable. Another relevant example from MNP pertains to a system of financial incentives (e.g., bonuses for hunters arrested, long distances walked) that reportedly improved patrol staff performance during a recent donor-funded project (GWD 2005), but had the unintended consequence of reducing staff morale when the project ended and incentives ceased. Such examples underscore the importance of careful planning for sustained improvements in program effectiveness.

Evidence for the success of the GWD monitoring program in closely involving local stakeholders and leading to rapid management decisions could also be seen as equivocal. While GWD officers and wildlife guards are intimately involved as local stakeholders in the conservation of park resources, formal involvement of broader stakeholders from communities around MNP (and other parks) has in the past been minimal or, in fact, adversarial in the sense of conflict between patrol staff and local hunters, or park wildlife and nearby farmers (e.g., Mason 1993; Sam et al. 2005). This has led to more recent promotion of collaborative management between parks and neighboring communities, including new initiatives for participatory monitoring of wildlife in community reserves (GWD 2000; Sheppard et al. 2010).

Within GWD, management and interpretation of monitoring has typically been “top down” and significantly influenced by outside experts (e.g., GWD 2005; Jachmann 2008a,b). The program does likely facilitate rapid management response, particularly with respect to anti-poaching efforts (e.g., patrol deployments to areas of recent hunting activity), though this is difficult to track with available data. Jachmann (2008b) suggests that feedback from the monitoring system has resulted in improvements in patrol staff performance and corresponding reductions in illegal hunting. Nevertheless, an explicit role for feedback from the monitoring program does not appear to be institutionalized, particularly with respect to wildlife data (as evidenced by minimal mention in a recent MNP management plan, GWD 2005).

In sum, these reflections suggest that the GWD program may not represent an ideal model of effective locally-based monitoring, despite its noted strengths. In fact, a simple dichotomy between locally-based and professional scientific monitoring systems may not be useful in practice since many programs combine elements of both and should be assessed based on their distinct features and the specific context in which they operate (Danielsen et al. 2009).

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