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Consumption of bushmeat around a major mine, and matched communities, in Madagascar

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Abstract

Mining can have serious biodiversity consequences and many mining operations take steps to mitigate their impacts. Evaluating their success poses a significant challenge because appropriate counterfactuals (what would have happened in the absence of the mine) are often unavailable. We aimed to estimate the effects of education and enforcement measures carried out by a large mine in eastern Madagascar on local consumption of illegal bushmeat. We adopt a quasi-experimental approach and use an interview technique designed to reduce sensitivity biases to compare levels of consumption amongst mine employees and people living within the mine's intervention area with those of statistically matched control groups, and to relate differences to respondents' knowledge of relevant wildlife laws. Consumption was lower, and awareness of the law higher, amongst mine employees and those living in the mine's intervention area. However caution should be applied in interpreting these results as evidence of the effectiveness of anti-bushmeat efforts by the mine due to potential confounding factors: for example abundance of bushmeat species may vary between the study areas, and our method may not have completely removed the sensitivity of questions about illegal consumption. This illustrates the challenges of evaluating conservation impacts. We highlight the low level of understanding of wildlife laws, including among mine employees, and suggest better communication of these laws, as part of an education programme, could be a useful first step towards reducing illegal hunting.

1. Introduction

The commercial extraction of valuable minerals is economically important in many parts of the world. Mining can have a positive impact on human development by generating jobs and raising government tax receipts (ICCM 2012), although Seagle 2012 and Filer 2006 discuss potential negative social impacts. However, mining can also have highly negative environmental impacts both directly, including through pollution (Uryu et al. 2001), habitat destruction, introducing alien species (Gould et al. 2011), and indirectly, by facilitating access for logging, agricultural expansion or hunting (Wilkie et al. 2008; Raiter et al. 2014). There is therefore a potential conflict between mining development, which may contribute to human wellbeing through economic growth, and biodiversity conservation, where the role of biodiversity in underpinning ecosystem services may also contribute to human wellbeing but be less well valued by markets.

To mitigate the potential negative consequences to biodiversity from mining activities, companies can adopt measures to minimise or prevent such impacts around mining areas. To minimise their negative effects, mines are often required by legislation, or the terms of their loans, to ameliorate their biodiversity impacts, and of course may go beyond national legislative requirements. Mitigation measures tend to follow a hierarchy: a) avoiding environmental impacts where possible, b) minimizing unavoidable impacts and c) remediating, offsetting or otherwise compensating for residual, negative effects (McKenney & Kiesecker 2010). Measures to mitigate the potential impacts of mining on biodiversity may include the designation of conservation areas and implementation of forest management plans, investment in alternative livelihoods, with the objective of taking pressure off remaining habitat, and education about and enforcement of conservation rules.

Madagascar possesses significant mineral resources (Cardiff & Andriamanalina 2007) and is also a global hotspot for biodiversity. In recent decades both artisanal and large-scale mining operations have increased across the country (Cartier 2009). Over the same period,

hunting of Madagascar's unique wildlife has come to the fore as a key conservation issue, with pressure on threatened and protected species linked to rising demand for wild meat and the breakdown of traditional taboos (Jenkins et al. 2011). Laws are a crucial aspect of conservation and natural resource management (Keane et al. 2008) and although Madagascar has a clear system of wildlife laws (Rakotoarivelo et al. 2011) which defines what species can be hunted, where and when, evidence suggests that these are often very poorly understood and therefore unlikely to influence behaviour (Keane et al. 2011). The major mines in Madagascar operating in biodiversity-rich areas attract significant international scrutiny and have made explicit commitments to reduce their net impacts on biodiversity (Vincelette et al. 2007; Ambatovy Project 2009) and reducing illegal hunting is a stated objective of Ambatovy Minerals and QIT Madagascar Minerals (QMM), Madagascar's two largest mines (Ramahavalisoa et al. 2012). Both Ambatovy and QMM use environmental education and enforcement measures as part of their strategies to minimise or offset their biodiversity impacts (e.g. Office Nationale de l'Environnement 2006), but the effectiveness of such efforts in changing behaviour has not previously been measured.

In this study, we aimed to evaluate the impact of the Ambatovy Minerals mine on the consumption of bushmeat in eastern Madagascar. In the absence of a controlled experiment, it is often difficult to draw robust conclusions about causality (Ferraro & Pattanayak 2006). It is therefore inherently challenging to investigate the impact of a major intervention such as a mine *post-hoc*; where the intervention is not placed randomly, adequate before and after comparisons do not exist, and the lack of replication of the intervention makes spatial comparisons problematic. For example, systematic differences (such as in terms of socio-economic variables) between the population exposed to the intervention and those not exposed could confound estimates of the intervention's true effect. Studying the impact of an intervention on potentially sensitive behaviour, such as bushmeat hunting, is particularly challenging as respondents may not be willing to admit to involvement, even if guaranteed anonymity (Solomon et al. 2007; St John

et al. 2010; Nuno & St John 2014). We therefore use a combination of specialized techniques to statistically reduce the potential biases caused by underlying systematic differences between our control and intervention samples (non-parametric matching; Abadie & Imbens 2011) and the reluctance of people to admit to illegal behaviour (the Randomized Response Technique, RRT; St John et al. 2012).

2. **Methods**

2.1 Study area

The Ambatovy mine, one of the world's largest lateritic nickel mines, started production in 2012 with operations planned to continue over a lifespan of 27 years. The mine itself is situated in an area of rainforest in eastern Madagascar (Figure 1) adjacent to the new protected area of Ankeniheny-Zahamena Corridor. The mine is connected to a refinery plant at Toamasina on the country's east coast via a 220km pipeline. The forest around the mine provides an important habitat for many globally threatened species, several of which are hunted for bushmeat (Goodman & Mass, 2010). The mine has committed to having a net positive effect on biodiversity by avoiding impacts where possible, minimizing unavoidable impacts, carrying out progressive footprint restoration and implementing a multi-component offset program (Ambatovy Project 2009). The mine's enforcement activities and environmental education among its staff and local villages form part of the forest management component of this program.

2.2 Data collection

Between February and June 2011 interviews on bushmeat consumption were conducted with three groups: mine employees (hereafter "employees"), people living in villages within the mine's zone of intervention but not employed by it ("intervention group") and people living in similar area outside of the mine's zone of intervention ("non-intervention group"). Both areas provide favourable conditions for agriculture, logging and hunting.

We sampled mine employees from a list provided by the mine administration, interviewing 30% of employees in each department. Villages from within the mine's zone of intervention were selected at random from the Ambatovy project databases. The area selected for comparison from outside the zone of intervention was in the commune directly north of the mine: an area with a similar level of access to forest and socio-economic setting. Villages in this area were selected at random, based on a Madagascar vegetation and habitation map (see Figure 1). In smaller villages (<30 households), we attempted to carry out interviews with every household; in larger villages we sampled households by following a zig-zag route and conducting interviews at every second or third household (cf. East et al., 2005).

Respondents were asked about their consumption in the preceding 12 months of 8 animal species (whose distributions include the study areas), and their knowledge of the legal status of each species (Table 1; Goodman & Mass 2010). Seven of the species are protected from hunting under Malagasy law while one is classified as a game species, so we used a specialised interview technique, the Randomised Response Technique (RRT), to reduce potential biases due to question sensitivity. The method had been extensively tested in both eastern and western Madagascar before being applied in this study (Razafimanahaka et al. 2012) and is useful for providing answers to sensitive questions of a yes/no format (i.e. it can give information on whether a species has been consumed, but not easily on the frequency or volume of consumption).

Pictures of the eight selected species, which had previously been tested locally to ensure they were easily recognised, were shown to respondents. The RRT survey followed a 'forced response' model (Lensvelt-Mulders et al 2005). Respondents were given a cloth bag with 10 balls (blue, white and black) in it. They were asked to take a ball from the bag (without looking) and not show it to the interviewer. They were asked to truthfully answer the question ('have you eaten this species in the last twelve months?') if they had chosen a blue ball (probability 8/10).

Respondents were asked to simply say ‘have eaten’ if they selected a white ball (P =1/10) and to say ‘haven’t eaten’ if they selected a black ball (P=1/10). Because the interviewer does not know whether a respondent is saying they have eaten a species because they have indeed eaten it, or because they selected a white ball, the interviewer does not have any definite information about the respondent. However, an unbiased estimate of the proportion of the population who have consumed bushmeat can still be obtained. We explained the method and said it was like a game (*kilalao*) and that like a game they should follow the rules. We then worked through two to four non-sensitive example questions (using pictures of fish, bush pig, snake and cow) depending on how quickly they appeared to understand the method and the protection it offered. The probabilities associated with each response are explained in full in Razafimanahaka et al. (2012). It is important to note that for species consumed infrequently, memories about whether consumption has occurred within the last twelve months may not be accurate. The team worked hard to remind respondents of important events which happened twelve months ago but a cautious interpretation would be to assume that some reported consumption may have occurred up to eighteen months before.

2.3 Ethical considerations

Interviews were carried out by VCR with field assistants (listed in acknowledgements). HJR and JPGJ attended some interviews and all interviewers were fluent in the local dialect of Malagasy. The research was conducted under Bangor University’s ethical framework. Informants were assured that taking parts in the interviews was voluntary and that all information would be anonymous (no individual identifiers were taken during interviews). The data collection method (RRT) ensured that sensitive information was not held about individuals, ensuring additional protection. The research, while conducted with permission and support of Ambatovy mine, was independent in that the mine were not involved in data collection or analysis.

2.4 Data analysis

When attempting to use non-experimental data to estimate the effect of an intervention there is a risk that analyses will be biased by the non-random allocation of individuals into "treated" and "control" groups. In this case, the assignment of villagers to the employees, intervention and non-intervention groups is likely to relate to differences in individual characteristics such as age, gender and education. In order to reduce the influence of potential selection biases we therefore adopted a quasi-experimental approach, using a non-parametric statistical matching technique to select appropriate control groups for each analysis such that the distributions of these individual covariates within the controls closely reflected those within the treatment groups (see supplementary material for full detail). Using the matched datasets, we estimated the differences in (a) consumption of each species and (b) knowledge of their legal status between people employed by the mine, or living within its zone of intervention, and the control groups using generalised linear models with binomial errors. For the consumption data, our models incorporated a specially adapted link function to correct for random noise introduced by the RRT procedure (St John et al. 2012).

2.5 Details of statistical matching

Data were analysed separately for each species. Matching was carried out on three variables: sex, age and level of education. As very few women were present in our sample of mine employees (6 out of 86) we removed all women from the dataset and matched on age and education alone for analyses estimating the effect of employment by the mine on bushmeat consumption. The matching technique employed uses a genetic search algorithm to find an optimal match between treatment and control groups via the bias-corrected matching method of Abadie & Imbens (2006). This procedure was implemented using the function "matchit" from the "MatchIt" package (Ho et al., 2011) in R version 2.14.0 (R Development Core Team, 2011). The genetic search algorithm was initialised with a population size of 1000 and matching was carried out with replacement, allowing each respondent to be matched more than once. Balance (i.e., the

success of matching) was assessed using Kolmogorov-Smirnoff tests for continuous variables and paired t-tests for dichotomous variables for each of the variables used to match the samples and all of their second order interactions. Matching with replacement means that data from respondents may be selected more than once, so subsequent analyses were appropriately weighted. The average effect of each "treatment" (being employed by the mine or living within its intervention zone) on the treated individuals was estimated with weighted generalised linear models using the "glm" function in R. For each species and treatment, two models were fit: one estimating the effect of the treatment on the proportion of individuals who had consumed the species in the previous year and the other estimating the effect on the proportion of respondents who believed that consumption of the species was legally permitted. Both of these response variables were binary, taking the values "yes" or "no". The data on consumption were collected using RRT, so for this analysis we used a glm with binomial errors and a specially modified link function which accounts for the stochastic uncertainty in the true value of the responses introduced by the RRT procedure (St John et al., 2012). The data on belief that consumption was permitted was collected using direct questions, so this analysis used a glm with binomial errors and the canonical logistic link function. Statistical significance of the treatment effects were calculated at the $\alpha = 0.05$ level using Wald t-tests.

Initially 313 mine employees were interviewed but this paper only includes data from the 86 employees who are local villagers and return home to their villages at night. The complete dataset therefore contained responses from 526 individuals (comprising the employees group, n = 86, the intervention group, n = 264, and the non-intervention group, n = 176) for 8 species, giving a total of 4208 responses. Prior to analysis, we removed all responses where individuals were unable to recognise the species in question from a photograph and a brief description (Table 2). The two tenrec species were both correctly recognised by 99% of the respondents. *Eulemur fulvus*, *Indri indri* and *Propithecus diadema* were recognised by between 80% and 88%, and *Cryptoprocta ferox* by 66% of respondents. By contrast, *Prolemur simus* was very poorly known

(only 26% of respondents recognised the species) so no further analysis was carried out for this species.

3. Results

First we compared bushmeat consumption in the previous 12 months by respondents living within the mine's zone of intervention (the intervention group) with a matched reference group drawn from those living outside the zone (non-intervention group; Figure 2a). These revealed that protected species were widely consumed in the matched reference group with the lemur *E. fulvus* consumed by 45 % of respondents, *A. laniger* by 42%, *P. diadema* by 36%, and the critically endangered *I. indri* by 17%. The carnivore *C. ferox* was consumed by 13%. The two tenrec species had been consumed by approximately two thirds of respondents (*H. nigriceps* 59%, *T. ecaudatus* 66%). Consumption amongst the intervention group (close to the mine) was significantly lower for all species except *I. indri* and *C. ferox*. The largest effect was observed for *E. fuvlus*, where the proportion of respondents who had consumed the species was 74% lower in the intervention group than in the matched reference group. However it is notable that the percentage of respondents who reported consuming protected species, even in this group, was very high with more than 20% reporting having eaten *A. laniger*, *P. diadema*, and *C. ferox*.

The majority of the matched reference group believed that consumption of the species included in the survey was legally permitted: 68% believed that consumption of *I. indri* was legal, while between 90% and 98% believed that consumption of the other protected lemur species was legal (Figure 2b). Belief that consumption is allowed was significantly lower in the intervention group, although the majority still believed that consumption was legal for *I. indri* with almost two-thirds believing it is legal to kill and eat the other three lemur species.

Our next analyses compared mine employees to a new matched reference group drawn from the non-intervention group. This group reflected the particular subset of the population from which mine employees are drawn (i.e., mine employees are overwhelmingly male and tend to be younger and more likely to have had a secondary school education than the population of the area

as a whole). Within this subset of the population, the proportion of people who had consumed lemurs and *C. ferox* were generally lower, whether individuals are employed by the mine or not (Figure 2c). However rates of consumption reached more than 25% for some lemurs (*P. diadema*). Both tenrec species were consumed by over half of respondents (*H. nigriceps* 56.7%, *T. ecaudatus* 64.0%). Consumption was consistently lower amongst mine employees than the reference. However, this difference was only statistically significant for the two game species: *H. nigriceps* and *T. ecaudatus*.

The majority of this new matched reference group believed that consumption of the species was permitted (61% for *I. indri* and between 90% and 100% for the other species; Figure 2d). Mine employment again had a large, significant effect for all species, with a 67%-79% reduction in the proportion of respondents who believed that consumption of the five protected species was legal compared to the control group. However despite this, more than 15-30% of mine employees believed that consumption of protected species was legal.

We explored the relationship between bushmeat consumption and knowledge of wildlife laws by plotting the proportion of individuals who had consumed each species against the proportion of individuals who believed consumption was permitted (Figure 3). These plots show a positive correlation between the two factors: when a smaller proportion of a group believe that a species can be consumed legally, levels of consumption also tend to be lower.

4. Discussion

4.1 Can the results be interpreted as evidence of reduced hunting caused by the mine?

Although our data do not contain information on the volumes of bushmeat consumed, they suggest that consumption for all the threatened species investigated is worryingly widespread. Indeed, the levels reported from the matched reference site were even higher than reported by Razafimanahaka et al. (2012). This adds to the growing body of evidence about the

threat that bushmeat hunting poses to Madagascar's endemic fauna (Jenkins & Racey 2008; Golden 2009; Randrianandrianina et al. 2010; Jenkins et al. 2011; Razafimanahaka et al. 2012).

Consumption of threatened species was found to be lower among people employed by the mine and those living within the mine's zone of intervention than amongst similar matched populations living in areas outside of the mine's zone of intervention. One interpretation of our results could be that the mine's education and enforcement activities have directly reduced bushmeat consumption in its zone of intervention. Further evidence in support of this interpretation comes from a previous study which found a decrease in the number of traps found in this zone (Ramahavalisoa et al. 2012). However, several other mechanisms could plausibly contribute to the observed differences so it should not be interpreted as conclusive evidence for a positive effect of the mine's activities.

First, although matching on observed characteristics can reduce selection biases arising from systematic differences between the populations under comparison (Abadie & Imbens 2011), only a few, relatively crude covariates (sex, age, level of education) were available here and other, unobserved characteristics might also have been important. The most obvious omitted variable was wealth. Wealth has previously been found to predict bushmeat consumption (East et al. 2005, Wilkie et al 2005, Godoy et al. 2010), though the relationship is neither straightforward nor consistent (Brashares et al. 2011). We chose not to include a measure of wealth in our matching procedure for two reasons. First, few simple, reliable indicators of wealth exist that are appropriate to the local context and could be recorded in rapid interviews of the sort employed here. Second, our study was initiated after the mining project had commenced and measures of participants' wealth prior to the creation of the mine were not available. Employment by the mine (or "trickle-down" to residents living close to the mine) is likely to affect wealth directly. Matching on "post-treatment" variables is undesirable and could have confounded our attempts to isolate the mine's effects. While we cannot entirely discount an effect of prior wealth in the

comparison between the group living close to the mine and those unaffected by it, we do not believe it was an important factor in explaining the observed differences in bushmeat consumption: we did not observe obvious, systematic differences in wealth between the villages within and outside of the mine's zone of intervention and a recent study found that wealth was not a particularly strong predictor of bushmeat consumption in a similar area of eastern Madagascar (Jenkins et al. 2011).

A second consideration when comparing bushmeat consumption by villagers from areas within and outside of the mine's zone of intervention, is that observed differences in consumption might also have been affected by spatial differences in the abundance of bushmeat species. For example, the abundance of lemurs, tenrecs and fossa might have been lower around the mine footprint than in the comparison site, either because of natural differences in abundance or differences in the management and exploitation of the areas. Surveys close to the mine have shown that the Indri is the least common species in the area, while the woolly lemur is the most common (Ralison 2010). Unfortunately, no comparable data are available for the area not directly impacted by the mine so it is not possible to assess the extent to which differences in abundance might contribute to differences in consumption between the two areas. However, it seems unlikely that mine employees and individuals from the intervention group would experience differences in species densities sufficient to result in the observed differences in consumption between them, since both groups come from the same villages.

Finally, while RRT was explicitly used to reduce potential biases associated with questions about illegal hunting, it is impossible to ascertain whether RRT was uniformly effective throughout the study area. If questions concerning illegal hunting were more sensitive amongst the groups exposed to the mine's activities (very possible given higher awareness of wildlife laws among these groups), such differences might have contributed to the reported differences in consumption and we consider this an important limitation of this research. RRT has been shown to reduce both non-response bias (where a non-random subset of potential respondents refuse to

participate in a survey; Lahaut et al. 2002) and social-desirability bias (where respondents mislead interviewers to present themselves more favourably), resulting in higher estimates of sensitive behaviour than conventional direct questioning which have been widely interpreted as evidence of more honest reporting (Scheers & Dayton 1987; Solomon et al. 2007; Silva & Vieira 2009). Where the true behaviour of respondents is somehow known, RRT returns more accurate responses than direct questions (Lensvelt-Mulders et al. 2005). Our methods built on an extensive trial of RRT as a tool to investigate illegal bushmeat consumption in Madagascar, during which we compared direct questions and RRT in nearly 1500 interviews across the country (Razafimanahaka et al. 2012). We are confident that the approach was understood by informants and that is did reduce the sensitivity of the questions. However, it is not possible to know whether it reduced sensitivity to the same level with both groups.

4.2 Explaining observed differences in bushmeat consumption

Our results suggest lower bushmeat consumption both amongst the mine's employees and within its zone of intervention. Several mechanisms could have produced this effect. For example, mine employees generally have higher incomes than those employed elsewhere. A recent study suggested that bushmeat is an inferior substitute for domestic meat in this region of Madagascar (Jenkins et al. 2011), so those employed by the mine may consume less bushmeat simply because they can afford preferable alternatives. We were unable to explore this hypothesis further in this study, instead focusing on the role of awareness of conservation rules. We found that awareness of wildlife rules was extremely low among the non-intervention references groups, with more than 80% believing that protected lemur species such as *E. fulvus* and *C. ferox* could be hunted and consumed legally, confirming the findings of the only previous study of villagers' knowledge of conservation rules in Madagascar (Keane et al. 2011). Knowledge of wildlife laws was higher among mine employees and villagers within the mine's zone of intervention and we found a positive relationship between the proportion of people who believe a species can be legally consumed and the estimated proportion who have eaten the species in the last year.

Tackling illegal bushmeat hunting is clearly a highly complex challenge. A number of species previously not heavily hunted (protected by local taboos and social norms; Jones et al, 2008) have become more extensively targeted as increased human mobility breaks down traditional natural resource management institutions, guns become more prevalent, and economic drivers change (Dunham et al. 2008, Barrett & Ratsimbazafy 2009, Jenkins et al 2011). However in some areas, bushmeat (including protected species) provides vital protein, contributing to human health (Golden et al. 2011). In 2011 the Malagasy government, in collaboration with academics and NGOs, developed a national bushmeat management strategy which lists the promotion of alternatives for bushmeat hunting as important activities, alongside increased communication and enforcement of wildlife laws. Our findings reinforce this message, as reveal that there is still limited understanding of wildlife laws in rural Madagascar, even among the employees of a major mine. Laws cannot be effective if they are not well understood (Keane et al. 2011) and improved communication could contribute to reducing hunting. Recent evidence from Madagascar (Rakotomamonjy et al. 2014) shows that relatively simple conservation education programmes can have a lasting impact on the knowledge and attitudes of participants for at least a year.

5. Conclusions

To achieve success, the conservation community should continually strive to evaluate the effects of conservation actions and adapt its strategies accordingly. Mines can have significant impacts on biodiversity, and accurately measuring the effectiveness of their mitigation measures is vital to ensuring that they are fulfilling their environmental obligations as well as their commercial and social commitments. We provide evidence suggesting that bushmeat consumption is lower among local people employed by a major mine than among the general local population, and that it is lower in villages exposed to the mines' interventions than with a matched sample of the population from similar, reference communities. Unfortunately, despite our best efforts to

overcome challenges of impact evaluation in this sensitive area using statistical matching and specialised methods for reducing sensitivity, it is not possible to conclusively conclude that these results are the result of the mine's activities. Our results highlight that in eastern Madagascar, many people perceive lemurs as a legal source of food, irrespective of whether they live near to, or work for, the Ambatovy mine. Of course ensuring people understand the law is not a guarantee of compliance, but it is an important first step and an area where further effort should be invested by any organisation seeking to reduce bushmeat consumption in its area of influence

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Table 1: Species included in the study showing their legal status and IUCN threat status (www.iucnredlist.org accessed on 14.10.14*). Hunting of species classified as protected under Malagasy law is entirely prohibited, while game species can only be hunted for personal consumption during certain months (Rakotoarivelo et al. 2011).*

Scientific name	*Common name*	*Legal status*	*IUCN status*
Indri indri	Indri	Protected	CR
Propithecus diadema	Diademed Sifaka	Protected	CR
Prolemur simus	Greater bamboo lemur	Protected	CR
Avahi laniger	Woolly lemur	Protected	VU
Cryptoprocta ferox	Fossa	Protected	VU
Eulemur fulvus	Brown lemur	Protected	NT
Hemicentetes nigriceps	Highland streaked tenrec	Protected	LC
Tenrec ecaudatus	Common tenrec	Game	LC

Table 2: Number of respondents who were correctly able to identify each species from a brief description and a photograph. The figures in brackets show this is as a proportion of the respondents' respective group. Responses from those who were not able to identify a species were discarded for the species in question, so these numbers represent the sample sizes for our analyses.

Species	Total (n = 526)	Mine employees (n = 86)	Intervention (n = 264)	Non-intervention (n = 176)
Avahi laniger	364 (69.2%)	69 (80.2%)	184 (69.7%)	111 (63.1%)
Eulemur fulvus	460 (87.5%)	75 (87.2%)	231 (87.5%)	154 (87.5%)
Tenrec ecaudatus	521 (99.0%)	86 (100%)	260 (98.5%)	175 (99.4%)
Hemicentetes nigriceps	520 (98.9%)	85 (98.8%)	259 (98.1%)	176 (100%)
Cryptoprocta ferox	346 (65.8%)	65 (75.6%)	184 (69.7%)	97 (55.1%)
Prolemur simus	134 (25.5%)	35 (40.7%)	74 (28.0%)	25 (14.2%)
Propithecus diadema	420 (79.8%)	78 (90.7%)	200 (75.8%)	142 (80.7%)
Indri indri	435 (82.7%)	74 (86.0%)	211 (79.9%)	150 (85.2%)

Figure 1: Map of the study area showing the mine activities and interview locations.

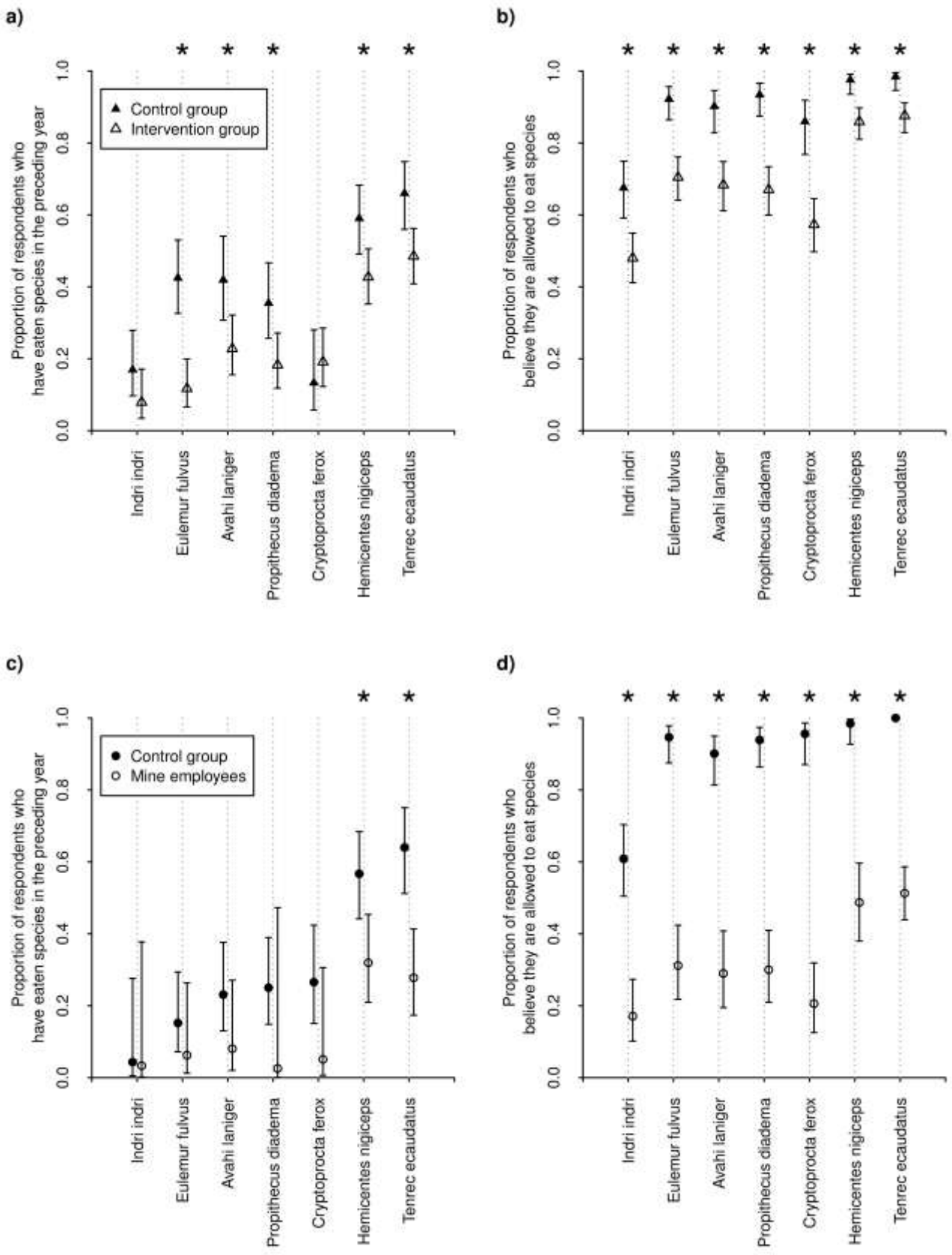

Figure 2: Comparisons between respondents living within the mine's zone of intervention

("intervention group") and a matched control drawn from those living outside the zone of intervention ("non-intervention group") (top row) and between mine employees and a matched control group drawn from the intervention group (bottom row). In each case, the figure compares the proportion of respondents estimated to have consumed each species in the previous year (left column) and the proportion of respondents who believe that it is legally permitted to consume them (right column). The mean estimated proportions for the control groups are marked by filled symbols and the mean estimated proportions for the "treatment" groups are marked by open symbols. Significant differences (at 5% level) between the control and treatment groups are indicated by an asterisk. Whiskers show 95% confidence intervals.

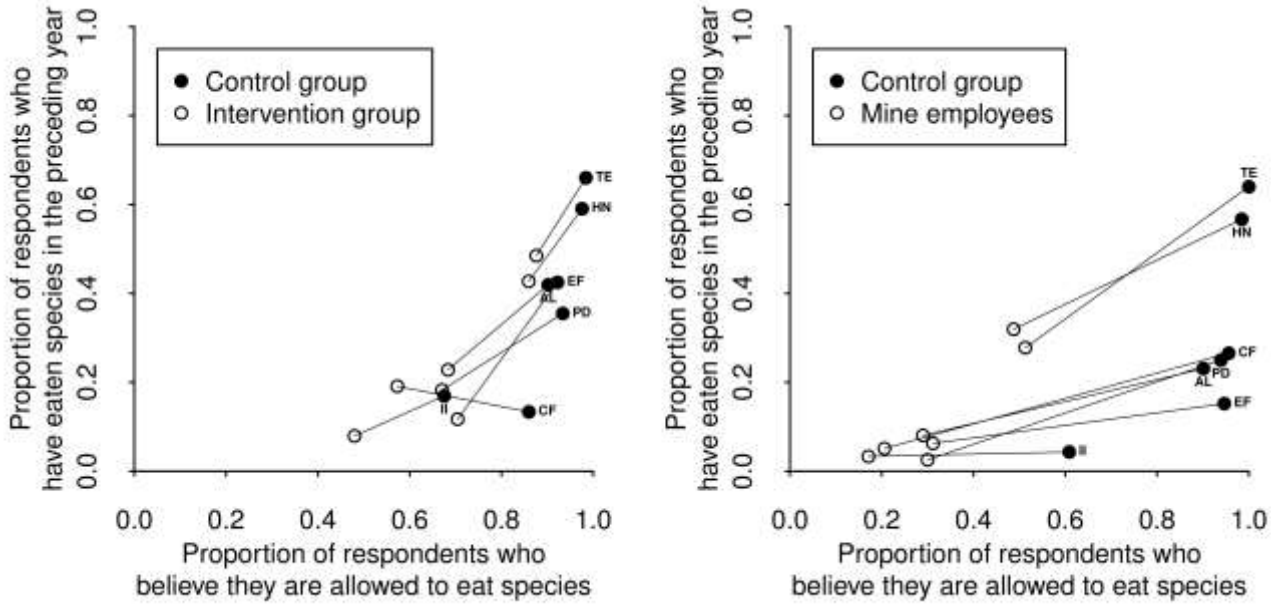

Figure 3: Relationship between the proportion of respondents who believe that it is legally permitted to eat a given species and the proportion of respondents estimated to have eaten the species in the previous year. The left panel compares respondents living within the mine's zone of intervention ("intervention group") to a matched control group drawn from those living outside of the zone of intervention ("non-intervention group") while the right hand panel compares mine employees against a control group drawn from the intervention group. In both cases, the "treatment" group is marked by filled circles and the control group is marked by open circles. Estimates for treatment and control groups for a single species are linked by a solid line. The species to which a pair of points refers is indicated by the following abbreviations: AL = Avahi laniger, *EF* = Eulemur fulvus, *II* = Indri indri, *PD* = Propithecus diadema, *CF* = Cryptoprocta ferox, *HN* = Hemicentetes nigriceps, *TE* = Tenrec ecaudatus.

Raw data: We have deposited the raw data (and metadata) used in this analysis under Elsevier's open data pilot.