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Controlling access to oil roads protects forest cover, but not wildlife communities: a case study from the rainforest of Yasuní Biosphere Reserve (Ecuador)

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Abstract

Through the analysis of a case study from Amazonian Ecuador, this paper evaluates the impacts of two oil-road management approaches on the structure and composition of wildlife communities (large- or medium-sized mammals and game bird species). In a free-access road, where forest has been cleared and fragmented by colonists, fewer species were found, together with wildlife density estimates that were almost 80% lower than on a control site without human disturbance. In contrast, on the road where access control has been enforced, habitat destruction has been minimal, but several wildlife species showed reductions in their populations, apparently related to changes in the subsistence practices of local Waorani hunters that settled along the road after its construction. In this area, economic subsidies and free transportation from the oil companies, access to the road, more efficient hunting technologies, and market incentives have increased the impacts of hunting by the Waorani, resulting in depletion of the local wildlife populations. Our research suggests that construction of roads in oil extraction areas must be avoided if at all possible. The alternative management of controlling access can be effective for short-term habitat protection, but not for wildlife conservation, especially when oil industry practices alter the social dynamics of local indigenous groups.

Introduction

Throughout the world, conservation of wildlife communities is challenged by an increasing overlap between human-dominated landscapes and the remnants of pristine habitats. While analyzing the impacts of this overlap, previous research has emphasized the more obvious and direct impacts of large human interventions such as expansion of the agricultural frontier, timber extraction, and the resulting loss and fragmentation of habitats (Lopes & Ferrari, 2000; Ochoa, 2000; Bonaudo *et al.*, 2005; Taylor & Goldingay, 2010). Because of their nature and magnitude, these human interventions leave little or no habitat available for wildlife species, with obvious effects on the composition, structure and abundance of their communities. In contrast, comparatively little attention has been paid to the *indirect effects* of other human activities (e.g. extractive industries) that could have dramatic long-term effects on wildlife communities, usually mediated by (1) changes in the accessibility to the landscape (i.e. roads); (2) changes in the subsistence practices of local human communities (Lu, 1999; Franzen & Eaves, 2007); and (3) alteration of the economic frameworks that govern the interactions between local people and wildlife (Thibault & Blaney, 2003; Godoy *et al.*, 2010).

As we consider the indirect effects of extractive industries in or near wildlife sanctuaries, oil and gas exploitation emerge as one of the most striking examples of the intricate conflicts arising from the overlap of wildlife refuges and other ecologically important areas, with the huge economic interests brought about by an ever-growing industry that support national economies. From the ecological devastation inflicted by the 2010 oil spill in the Gulf of Mexico (Campagna *et al.*, 2011, DeLaune & Wright, 2011; Richards *et al.*, 2011), to the social and legal uproar around the trial against Chevron (TEXACO) over its alleged responsibility for the pollution of a large tract of the Ecuadorian Amazon (Olsen, 2001; Yoder, 2002; Valdivia, 2007; Finer *et al.*, 2009), the history of oil industry operations in ecologically and culturally sensitive areas is plagued by cases of environmental pollution and ecosystem destruction, social unrest and a sense of opportunities lost, among those who see potential benefits in collaborations between conservation and industry. But in the midst of the rocketing energy demands worldwide, and the inevitable expansion of the oil industry, research efforts have focused on the assessment of direct impacts of oil spills or other forms of pollution on wildlife and ecosystems, while little is known about the effects of different management alternatives which could mitigate some of the impacts of the activities and infrastructure related to oil. By using a case study from Yasuní Biosphere Reserve (YBR), in this paper we discuss the differential impacts of two types of management implemented on roads opened to allow oil exploitation in the Ecuadorian Amazon. In particular, we evaluate the effects of road access control in terms of protecting wildlife communities (large- or medium-sized mammals and game bird species) in lowland neotropical rain forests in Ecuador.

The effects of roads have been widely analyzed with regards to their direct impacts on wildlife and habitat destruction (Forman & Alexander, 1998; Spellerberg, 1998; Jones, 2000; Goosem, 2002; Clevenger, Chruszcz & Gunson, 2003; Greenberg *et al*., 2005; Coelho, Kindel & Coelho, 2008; Laurance, Goosem & Laurance, 2009; Benítez-López, Alkemade & Verweij, 2010; Taylor & Goldingay, 2010). Road construction has immediate impacts on wildlife mobility and mortality, and direct effects on forest cover, which are usually small in terms of total area impacted. More importantly, roads provide access to previously remote regions, thus facilitating colonization, deforestation, exploitation of wildlife and agricultural encroachment (Nepstad *et al*., 2001; Bonaudo *et al*., 2005; Laurance *et al*., 2009; Southworth *et al*., 2011). Furthermore, roads usually promote commercial exchange among newly connected areas, accelerating habitat destruction and further intensifying hunting and its impacts on wildlife (Suárez *et al*., 2009; but see Ayres *et al*., 1991). In the rainforests of Gabon, for example, hunting associated with the presence of roads reduced population abundance and changed the behavior of several large-bodied mammals (Laurance *et al*., 2006). Similarly, proximity to the access matrix (roads and rivers) in the Brazilian Amazon has been shown to predict the abundance of game species, whose abundances were depressed in areas that are closer to these transportation axes (Peres & Lake, 2003). While these impacts are well documented, there is virtually no information about factors, such as access and road-use restrictions, that could help mitigate some of their negative impacts.

The Ecuadorian Amazon, with its exceptional levels of wildlife diversity, the presence of sensitive indigenous groups, and the influence of an active oil industry, provides an excellent setting to explore the interactions that emerge in areas where cultural and ecological integrity are confronted with the changes that the oil industry bring. At the same time, given the heterogeneity of oil exploitation approaches and policies that have been implemented, this area is ideal to explore mechanisms or practices that could minimize the negative impacts that this economic alternative has on native ecosystems and human communities. In particular, YBR offers an unusual opportunity to consider mechanisms or practices that could minimize the negative impacts of oil exploitation on native ecosystems and human communities. In a previous paper (Suárez *et al*., 2009), we explored the structure of a wild-meat market that emerged in the YBR, and its relationship with the roads and the operation of oil companies in this region. At that time, however, we did not have enough information to assess the effects of this market on the local wildlife communities. In this paper, we discuss new information on the impacts of these roads by comparing the efficacy of road control policies in terms of reducing deforestation and wildlife depletion in natural areas of tropical lowland forests in eastern Ecuador. With this objective, we discuss the results of wildlife surveys in the area of influence of two contrasting roads in the YBR: the 'Auca' road, which is an open-access road built in the early 1970s, and the 'Maxus' road, which is a controlled road built in 1992 into the core of this protected area.

Methods

Yasuní Biosphere Reserve (~76°00'W, 01°00'S) is located in the Ecuadorian Amazon at elevations ranging from 200 to 500 m (Fig. 1), and has been recognized as a globally important area for biodiversity conservation (Bass *et al*., 2010). The reserve, composed of Yasuní National Park (YNP) and the contiguous Waorani Ethnic Reserve, encompasses an area of approximately 18 000 km^2, with warm climate and 3500 mm of annual precipitation typically described as aseasonal. The landscape is dominated by *terra firme* forest divided by small areas of floodplain, swamp, and successional forest (Balslev & Renner, 1989).

To evaluate the effects of two oil-road management approaches, we focused our research in three areas of the YBR (Fig. 1): (1) Taracoa, located on one of the branches of the Auca Road; (2) Yasuní Scientific Station, located on the southern bank of Tiputini River, at km 52 of the Maxus Road; and (3) a control site without recent human disturbance, located along the Tiputini River. The maximum distance between these areas is roughly 89 km, and they are all dominated by *terra firme* forests with uniform structure and composition. These sites, however, differ in the amount of disturbance that they have experienced: the Auca road was built in 1972 to allow oil exploitation in the region to the west of YNP, and triggered an uncontrolled flood of colonists, resulting in deforestation rates of about 4% per year between the 1970s and 1990s (Wunder, 1997), and an approximate forest loss of 40% in the vicinity of the road (GeoPlaDes, 2010). In contrast, the 140-km Maxus road was built in 1992 to allow oil exploitation inside YBR itself. Before the road's construction, the only significant impacts on this area were occasional subsistence hunting events by the Waorani groups that occupied the region. However, the low densities of these groups and their semi-nomadic customs suggest that these hunting practices had a negligible impact on wildlife communities (Yost & Kelley, 1983; Vickers, 1988). After construction of the Maxus road, the oil company established a control policy, which limited the

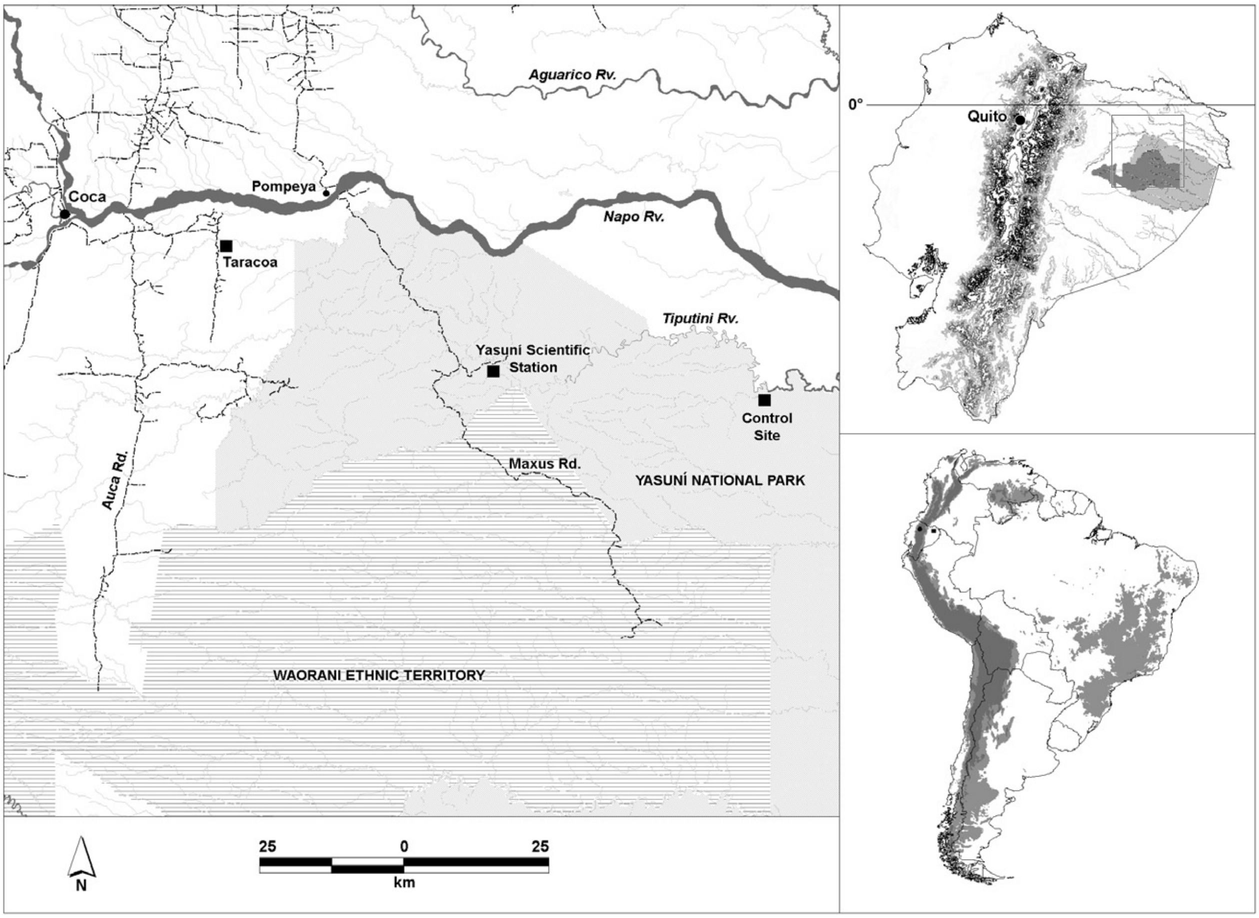

Figure 1 Map of the study area, including all the localities included in the text. Only the northern section of Yasuní National Park and the Waorani Ethnic Reserve are shown in the inset map.

access of outsiders and the impacts of colonization. The new road, however, attracted local Waorani people who settled along the road and use it as a hunting corridor. Thus, while the control strategy in this road was effective in terms of avoiding colonization and deforestation (until 2005, less than 2% of the forest had been lost (Greenberg *et al.*, 2005)), cultural changes among the Waorani and the transportation subsidies that they receive from the oil company, turned them into major suppliers of bushmeat to the market located at the origin of the road (Suárez *et al.*, 2009).

To estimate the density of wildlife populations, from April 2005 to July 2006, we carried out a series of bi-monthly surveys of large birds (several cracid species, toucans, and the grey-winged trumpeter), and medium-sized and large mammals (>1 kg). Surveys were performed using distance sampling techniques along six line transects (2 km each) in each of the three sampling sites (Buckland *et al.*, 2001). Each transect originated and continued perpendicularly from the roads (Taracoa and Yasuní Research Station) or from the river (Control site).

Each transect was surveyed during the morning (06:00–08:00 h) and evening (18:00–20:00 h) during eight time periods, by teams of three people walking slowly (~1 km h^{-1}) and recording direct wildlife observations. To avoid violating the assumption of certain detection near or close the transect line, observers paid particular attention to these areas, especially for birds and primates potentially hidden by thick foliage. To standardize efforts, surveys were not conducted during rainy days (Peres, 1999). For each observation, we recorded the species, number of individuals in the group, as well as sighting radial distance and angle to the center of the group (attempts were made to obtain these measurements accurately before any responsive movement occurred to avoid biased estimates of density and abundance). The analysis was conducted using Distance 5.0 Beta 3 (Thomas *et al.*, 2010). Population densities were derived by applying uniform and half-normal key functions, and cosine or simple polynomial adjustment terms as needed. Because of low encounter rates for most species, data were pooled over the eight time periods according to taxonomic ranks (order) to fit detection functions (with further pooling across orders for some of the bird species). Goodness of fit tests were used to identify violations of assumptions. Akaike's information criterion was used in model selection, with particular attention paid to model fit at distances near zero because the fit of the shoulder near zero is most important for robust estimation (Buckland *et al*., 2001). Encounter rate was estimated separately for each site by species, but pooled over time. A Poisson distribution was assumed to estimate the variance in encounter rate. Group size was estimated separately for each species pooled over time and sites, unless analysis of variance indicated that group size varied significantly by site, in which case site-specific estimates were used. Average group size was used, unless the test for size bias at the 10% level of significance (determined by means of a regression of the natural logarithm of group size against distance from the transect line) indicated that expected group size was more appropriate. Density of groups and of individuals by species for each of the three sites was estimated. Biomass for each species was calculated using average density estimates derived from our surveys, which were multiplied by the mean body weight reported in the literature for each species (Emmons, 1999).

Results

After a total sampling effort of 254 km, we recorded 268 individuals belonging to 24 species (sighting rate: 1.06 sightings per km). Large variation in sample size was observed among species, from one observation each for five species (*Alouatta seniculus*, *Saguinus fuscicollis*, *Nothocrax urumutum*, *Ortalis guttata* and *Cuniculus paca*) to 32 observations for *Sciurus* spp. Even pooling the data by order (or across orders for some of the bird species), after right truncation to improve model fit, we only reached the upper limit of the recommended sample size for primates, just over half that limit for the bird and rodent species, and under a quarter for Artiodactyla (see Fig. 2 for details of each detection function fit). Even so, the detection functions were reasonable with the possible exception of that for Artiodactyla.

Species composition and abundance exhibited important differences. Overall, in the highly impacted area along the Auca road, we recorded fewer species than in the other two sites, and wildlife densities one half that of the Maxus road, and one quarter that of the control site (Table 1). Among the most notable species that were absent from our surveys on the Auca road are three primate species previously reported for this region (*Ateles belzebuth*, *Lagothrix lagotricha* and *A. seniculus*), and the white-lipped peccary *Tayassu pecari*. In the case of the Maxus road, species composition in this area was relatively similar to that in the control site, with the exception of the absence of the white-lipped peccary, Salvin's curassow *Mitu salvini* and the speckled chachalaca *O. guttata*, which were not directly recorded along the transects in this area. However, our density estimates for most of the species was much lower on the Maxus road (Table 1), resulting in total wildlife densities that were 2.4 times lower at the Maxus site (110 individuals per km^{-2}) than at the control (265 individuals per km^{-2}). Of the 24 species reported, 11 exhibited higher average densities in the control that within the Maxus or Auca road sites, whereas only four species were more abundant along the Maxus road than in the control site (Table 1).

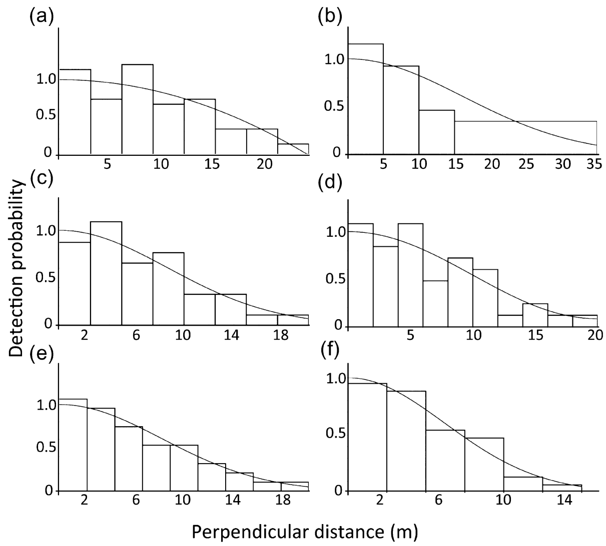

Figure 2 The detection function superimposed on histograms showing the frequency of counts in each distance interval with data pooled by order for the (a) primates – uniform key function with polynomial adjustment terms with right truncation at 25 m; (b) Artiodactyla – half-normal with no adjustment terms with right truncation at 35 m and data grouped into four intervals for analysis; (c) Galliformes – half-normal with no adjustment terms with right truncation at 20 m; (d) Galliformes + Gruiformes – uniform with cosine adjustment terms with right truncation at 20 m; (e) Galliformes + Gruiformes – (f) Rodentia – half-normal with no adjustment terms with right truncation at 15 m.

Differences among faunal communities in the three sites are even more striking when analyzing wildlife biomass estimates. At the control site, we estimated a total biomass of 4380 kg km^{-2}, with almost 87% being contributed by the white-lipped peccary (Table 2). However, even if we exclude this large-bodied and gregarious species from the analysis, the control wildlife biomass estimate (509 kg km^{-2}) was 1.7 times greater than on the Maxus road (295 kg km^{-2}), and six times greater than in the Auca road site (84 kg km^{-2}). Most of this biomass decline is due to smaller populations of large-bodied monkeys, ungulates and game bird species.

Discussion

Although line transects have been used in several neotropical localities (Carrillo, Wong & Cuarón, 2000; Peres, 2000b; Cullen, Bodmer & Valladares-Padua, 2001), when encounter rates are low, this sampling technique can require considerable survey effort to obtain adequate sample sizes to fit detection function models (Peres, 1999; Buckland *et al*., 2001). Although, our sample sizes were relatively small, our density estimates lie in the same range as those from other neotropical areas (Hill *et al*., 1997, Peres, 2000b; Cullen *et al*., 2001). Given our relatively small sample sizes, our discussion emphasizes not the overall density or biomass estimates for each species, but the relative differences among

Table 1 Population density estimates for wildlife species recorded though direct observations along line transects established in the area of influence of two oil extraction roads (Auca and Maxus), and a control site in Yasuní Biosphere Reserve

	Control		Auca road		Maxus road	
	D	95% CI	*D*	95% CI	*D*	95% CI
Primates						
Alouatta seniculus	0	–	0	–	0.825	0.161–4.220
Ateles belzebuth	21.476	8.549–53.953	0	–	6.534	1.805–23.649
Callicebus moloch	7.46	3.886–14.323	0	–	3.095	1.044–9.171
Cebus albifrons	9.766	4.189–22.768	7.982	2.652–24.028	4.952	1.368–17.923
Lagothrix lagotricha	17.253	9.545–31.187	0	–	17.496	8.664–35.331
Pithecia monachus	1.809	0.617–5.300	3.942	1.518–10.239	1.834	0.517–6.504
Saguinus fuscicollis	0	–	0	–	0.825	0.161–4.220
Saguinus tripartitus	12.193	5.652–26.306	11.39	4.345–29.858	21.197	10.221–43.957
Saimiri sciureus	28.621	13.767–59.498	11.697	3.276–41.767	32.652	14.383–74.122
Artiodactyla						
Mazama americana	0.608	0.156–2.374	0	–	0.462	0.090–2.364
Pecari tajacu	2.448	0.809–7.408	0.8	0.156–4.091	0.745	0.146–3.807
Tayassu pecari	115.597	41.984–318.275	0	–	0	–
Rodentia						
Cuniculus paca	0.586	0.115–2.998	0	–	0	–
Dasyprocta fuliginosa	2.345	0.908–6.059	1.917	0.542–6.776	0	–
Microsciurus flaviventer	0.586	0.115–2.998	0	–	0.892	0.174–4.560
Myoprocta pratti	1.173	0.332–4.145	1.917	0.542–6.776	0.892	0.174–4.560
Sciurus spp.	11.214	6.699–18.771	8.147	3.998–16.602	5.686	2.548–12.688
Galliformes						
Mitu salvini	11.372	5.669–22.812	0	–	0	–
Nothocrax urumutum	0	–	1.377	0.269–7.041	0	
Ortalis guttata	0.421	0.082–2.154	0	–	0	–
Penelope jacquacu	10.108	5.637–18.125	2.361	0.663–8.403	3.295	1.119–9.702
Pipile pipile	6.706	3.299–13.635	3.654	1.229–10.868	1.133	0.222–5.796
Gruiformes						
Psophia crepitans	2.205	0.603–8.061	1.802	0.352–9.215	5.031	1.650–15.343
Piciformes						
Ramphastos spp.	1.029	0.285–3.707	0	–	3.13	1.175–8.336

The two roads differ in their management approach from open access and no control for the Auca road, to controlled access for the Maxus road. *D*, density of individuals per km; CI, confidence interval.

the sites in terms of species composition and estimated density and biomass, derived from equal sampling efforts and survey methods.

The purpose of this study was to use a case study in Amazonian Ecuador to assess if controlling access to roads in oil extraction areas was enough in terms of avoiding detrimental impacts on tropical wildlife communities, with emphasis on large- or medium-sized mammals and game bird species. Although access control to the Maxus road in the YNP has been shown to be an effective strategy in terms of restricting colonization, forest loss and habitat fragmentation [only between 2 and 7% of the area has been deforested in the vicinity of the road (Greenberg *et al*., 2005; GeoPlaDes, 2010)], our data suggest that wildlife has been severely impacted in this area, underscoring the importance of the indirect effects associated with the presence of roads and the oil industry.

The lack of fauna at the Auca road site was no surprise, as this area has been subjected to forest clearing and fragmentation, and intense hunting since its construction. In this area, open access to the roads resulted in rapid colonization and the ensuing deterioration of wildlife communities, as evidenced by the apparent absence of at least 11 species, never observed during this study. Interestingly, the species lost along the Auca road include not only hunting-sensitive species, but also small species such as titi monkeys, squirrels and toucans, whose absence from this area probably reflects habitat loss rather than over-hunting. Other studies that have assessed the effects of roads, habitat destruction and hunting have shown similar results (Cullen, Bodmer & Valladares Padua, 2000; Peres, 2001; Riley, 2002; Naughton-Treves *et al*., 2003; Kumara & Singh, 2004; Michalski & Peres, 2005; Laurance *et al*., 2006; Laurance *et al*., 2009; Parry, Barlow & Peres, 2009; Rovero *et al*., 2012).

In dramatic contrast with the Auca road, the area of influence of the Maxus road exhibits little signs of habitat loss and fragmentation, emphasizing the effectiveness of restricting access to oil extraction roads, in terms of avoiding human colonization and its accompanying processes of habitat deterioration. In spite of this, wildlife communities in this area have been impacted as suggested by the absence or lower abundance and biomass of several species. The

Table 2 Average biomass estimates of wildlife recorded though direct observations along line transects established in the area of influence of two oil extraction roads (Auca and Maxus), and a control site in Yasuní Biosphere Reserve

Species	Biomass/ individual (kg)	Total biomass (kg) per km^{-2}		
		Control	Auca road	Maxus road
Tayassu pecari	33.50	3872.5	0.0	0.0
Ateles belzebuth	8.00	171.8	0.0	52.3
Lagothrix lagotricha	8.00	138.0	0.0	140.0
Pecari tajacu	20.00	49.0	16.0	15.0
Mitu salvini	3.10	35.3	0.0	0.0
Cebus albifrons	2.50	24.4	20.0	12.4
Saimiri sciureus	0.80	22.9	9.4	26.1
Mazama americana	22.10	13.4	0.0	10.2
Dasyprocta fuliginosa	3.60	8.4	6.9	0.0
Penelope jacquacu	0.75	7.6	1.8	2.5
Callicebus moloch	1.00	7.5	0.0	3.1
Sciurus spp.	0.57	6.4	4.6	3.2
Saguinus tripartitus	0.50	6.1	5.7	10.6
Pipile pipile	0.75	5.0	2.7	0.8
Cuniculus paca	7.80	4.6	0.0	0.0
Pithecia monachus	2.50	4.5	9.9	4.6
Psophia crepitans	1.20	2.6	2.2	6.0
Myoprocta pratti	0.71	0.8	1.4	0.6
Ramphastos spp.	0.75	0.8	0.0	2.3
Ortalis guttata	0.50	0.2	0.0	0.0
Microsciurus flaviventer	0.09	0.1	0.0	0.1
Alouatta seniculus	6.00	0.0	0.0	5.0
Saguinus fuscicollis	0.50	0.0	0.0	0.4
Nothocrax urumutum	2.50	0.0	3.4	0.0
Total		4381.94	83.88	295.25

The two roads differ in their management approach from open access and no control on the Auca road, to controlled access on the Maxus road.

reasons for this decline are probably complex, and we can only suggest potential explanations. However, it is interesting that most of the species that contribute to this decline are common target species of the Waorani hunters who live along the Maxus road and include white-lipped and collared peccaries, spider and capuchin monkeys, and Salvin's curassow, Spix's guan *Penelope jacquacu*, and common piping guan *Pipile pipile*. These species have been shown to contribute between 54 and 84% of the biomass consumed by Waorani families living along this road (Franzen, 2006). In this context, our data suggest that the wildlife decline along the Maxus road could be related to the intensification of hunting by the Waorani. This conclusion is further supported by the importance of these species in the market of the Kichwa community of Pompeya, which receives wild meat from communities along the Maxus road and the Napo River. Between 2005 and 2007, for example, peccaries and spider monkeys accounted for 55% of the total biomass and 48% of the animals that were sold in this market (Suárez *et al*., 2009). The depletion of wildlife along this controlled road has been already suggested on the basis of hunting profiles, which showed that the populations of spider monkey *A. belzebuth* and howler monkey *A. seniculus* were experiencing severe depletion in areas influenced by older Waorani communities along other segments of the Maxus road (Franzen, 2006).

Exceptions to this pattern were the woolly monkey and the howler monkey; despite being important species in the Waorani diet, these species were as abundant or even more abundant in the Maxus road site than in the control site. We attribute this difference to the fact that two of our transects along the Maxus road ran through a primate research plot in which monkey hunting by the Waorani was restricted through an economic subsidy provided to Waorani families by a research project. Thus, regarding these species, our data on the impact of the Maxus road are confounded by the effects of this conservation subsidy focused on these two species. For the remaining large species, our data from the Maxus road resemble patterns found by previous research comparing Amazonian sites exposed to contrasting levels of hunting, where a decrease of wildlife biomass has been reported in heavily hunted areas (Cullen *et al*., 2000; Peres, 2000a; Peres & Dolman, 2000; Bonaudo *et al*., 2005; Endo *et al*., 2010), similar to the 42% biomass decline that we estimated for the Maxus road site.

The depletion of wildlife populations in the relatively intact forests along the Maxus road, recalls the 'empty forest' syndrome (Dirzo & Miranda, 1990; Redford, 1992; Wilkie *et al*., 2011). According to this pattern, dramatic ecological changes could be expected in this area as a result of the decline of several species that play key roles in the dynamics of the vegetation such us peccaries, monkeys and several species of cracids. From this perspective, our results suggest two important conclusions regarding the management and control approaches used for roads in oil extraction areas. On one hand, the control of access to newly opened roads can be very effective in terms of avoiding colonization and the ensuing changes in land-use that result in forest loss and fragmentation, as can be seen while comparing remaining forest cover along the Auca and Maxus roads. Moreover, the lack of an impact on forest cover suggests that other groups of non-hunted wildlife (e.g. small birds, amphibians) could benefit from road access control. On the other hand, it is clear that access control is not enough to protect local populations of large- and medium-bodied vertebrates, especially when the presence and operation of the road, in combination with the practices of the oil companies, promote drastic changes in the living conditions and subsistence activities of local people. In the specific case of the Waorani who live along the Maxus road, it has been shown that the economic and transportation subsidies from oil companies increase their participation in the bushmeat trade and expand the area influenced by their hunting practices (Suárez *et al*., 2009). Together with the adoption of more effective hunting technologies, this change is the most likely explanation for the severely depleted wildlife community that we found in this area.

The conclusions that we have outlined in this paper could be questioned in two important aspects: First, the probability that the differences in wildlife communities reported here could result not from the contrasting management approaches of the two roads, but from the very different age of the two roads. Second, the impossibility of finding true replicates to ensure that our results are not just a particularity of the system that we studied. Although the first possibility, cannot be discarded just based on our data, land-use analysis in this area (GeoPlaDes, 2010) suggest that this might not be the case: while the Auca road exhibits a forest loss rate of roughly 4% per year since its construction, the Maxus road has experienced a rate of only 0.33% per year. Moreover, the forest loss pattern at the Auca road exhibits the typical 'fish-bone' configuration that has been reported in other Amazonian forests where non-indigenous colonists use new roads to establish farms in pristine areas (Skole & Tucker, 1993; Sierra, 2000; Locklin & Haack, 2003; Oliveira de Filho & Metzger, 2006). In contrasts, the limited forest loss reported along the Maxus road is restricted to the immediate surroundings of the road axis. While the two areas are occupied by Waorani communities, the Auca road also experienced rapid occupation by non-indigenous colonist from other parts of the country and an ample expansion of the road system, two differences that can only be explained by the control regime implemented by the oil companies in the Maxus road, and the absence of similar measures in the Auca road.

Regarding the second limitation of our study, the lack of true replicates and the validity of the resulting conclusions, this is a factor that has been previously discussed in the context of wildlife studies (Johnson, 2002). In this regard, we support the idea that a different paradigm is needed while judging the validity of wildlife and conservation studies. In very real terms, the particularities of the social and ecological contexts mean that it is virtually impossible to find true replicates for this kind of research. In our case, for example, even if we could find an additional pair of roads with the same management approaches that we analyzed in the YBR, differences in the ethnic groups inhabiting those areas, the social or economic drivers that promote colonization, and even the community-relationship frameworks used by different oil companies would render any strict statistical comparison irrelevant. The uniqueness of these case studies, however, does not mean that we cannot learn from them. In fact, we argue that, by documenting these case studies, we provide perspectives and highlight the challenges that are likely to arise as wildlife managers and decision-makers are confronted with their own 'unique' ecological and socioeconomic settings. From this point of view, we believe that wildlife management in these complex systems can be informed more efficiently by the publication and analysis of several case studies (meta-replication; Johnson, 2002), that from the execution of strictly replicated studies that will demand armies of researchers and very long research times that we cannot afford.

In the context of increasing conflict that will emerge as ecologically sensitive areas are targeted for access to economically valuable resources such as natural gas, oil or minerals, we suggest that the conservation of wildlife and its habitats will depend heavily on decisions concerning the construction and management of roads. The control of public access to oil roads can reduce the colonization of new areas and the ensuing deforestation. Moreover, we cannot discard the possibility that this practice may even protect plant communities and non-game wildlife species. However, our results suggests that, given the indirect effects of oil industry on the economy and cultural practices of local people, these roads can have significant impacts on wildlife and on the ecological processes of their ecosystems. In this context, we conclude that, if at all possible, construction of new roads in 'pristine' areas should be avoided, as the most effective way to reduce the risk of colonization and the long-term costs that would be associated with the control of newly opened roads in ecologically or culturally sensitive areas. If road construction is indeed unavoidable, access control policies should be implemented, but could be insufficient in terms of precluding severe impacts on wildlife, especially in areas where the operation of new roads and the interactions between local people and oil companies affect the subsistence practices of local communities and their relationships with local wildlife and ecosystems. From this point of view, our study points out the urgent need to explore more responsible practices and regulations for the interactions between local communities and industry, in which the indirect effects (e.g. access to markets, improved transportation, cash economy) are taken into account and evaluated explicitly regarding their impacts on wildlife.

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