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Road mediated spatio-temporal tree decline in traditional agroforests in an African biosphere reserve

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ABSTRACT

Understanding the role of protected areas in conserving biodiversity is a central goal in conservation biology. Anthropogenic activities around and inside these protected areas and, in particular, roads can alter the spatiotemporal dynamics of biological diversity in protected areas. However, our understanding of how the presence and position of roads affect human attitude, subsequent agricultural practices and biodiversity conservation is limited. In this study, we tested the effects of the proximity of traditional agroforestry parklands to main roads used by park rangers for surveillance on the diversity and abundance of woody species in the Pendjari Biosphere reserve over 16 years (2000–2016). Tree density in agroforestry parklands decreased over time from an average of 20 trees/ha in 2000 to 7 trees/ha in 2016. Species such as *Vitellaria paradoxa* and *Parkia biglobosa*, which are economically important, experienced the largest density reduction Trees density was also significantly higher in farms close to the monitoring roads used by park rangers to patrol the park. Farms that are far from the roads were monitored less frequently given that the number of park rangers declined over time. However, there was no significant variation in species richness and diversity over time, perhaps because of the low tree diversity in these systems. This masks evidences of species local extinctions. For example, species such as *Pterocarpus erinaceus*, *Anogeissus leiocarpa*, and *Burkea africana* which were present in the traditional agroforestry parklands in 2000 disappeared by 2016. This is associated with important land-use changes including the conversion of gallery forests into cropland and wooded savannas indicating that human pressure not only affects species occurrence but also their habitats. Our study suggests that where land demand for agriculture is high, it is challenging for local people to maintain sustainable management practices in the absence of collective action.

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1. Introduction

Understanding the role of protected areas in conserving biological diversity is a central goal in conservation biology (Gaston et al., 2006; Le Saout et al., 2013; Venter et al., 2018). Protected areas are expected to function as a single conservation strategy (Geldmann et al., 2013). However, with rapidly increasing world population, land shortage in rural areas and pressure from agricultural intensification (Hadush et al., 2019; Laurance et al., 2012; Tilman et al., 2001), the efficiency of protected areas in conserving biodiversity has been questioned (Chape et al., 2008; DeFries et al., 2005; Houehanou et al., 2013; Mora and Sale, 2011). For example, 55% of African protected areas are inefficient at biodiversity conservation (Muhumuza and Balkwill, 2013). In East Africa, only half of national parks and forest reserves, and 8% of ranches are effectively conserved (Pfeifer et al., 2012). Less than 50% of bryophytes (Silva et al., 2014; Vergilio et al., 2016) and less than 50% of vascular plants, mollusks, arthropods and vertebrates are efficiently conserved in the protected areas of Azores (Vergilio et al., 2016). Under such persistent evidence that protected areas may not always conserve biological diversity, it is critical to further our understanding of the conditions under which protected areas can deliver the biodiversity conservation expectations (Geldmann et al., 2013). Local people who live around these protected areas play a central role. Understanding the spatiotemporal dynamics of the attitudes and practices of local people around these areas can shed light into why some protected areas succeed while others fail to deliver these conservations outcomes (Agrawal and Redford, 2009; Bare et al., 2015; Vodouhê et al., 2010).

Biosphere reserves are important biodiversity reservoirs but also a source of non-timber forest products (NTFP) which provides income for neighboring people (Gray et al., 2016; Joppa et al., 2008; Naughton-Treves et al., 2005). These reserves are established to ensure the long-term conservation of nature, ecosystem services and are linked cultural values (Dudley, 2008). Because biosphere reserves are also designed to provide economic and social benefits to local people while ensuring the conservation of biodiversity, they are structured into three zones with different functions (Batisse, 1985): (i) a core zone, which is strictly protected and where only research and monitoring activities are allowed, (ii) a buffer zone used for low impact tourism, forestry, and agriculture in line with overall conservation objectives and (iii) the transition zone, often the largest, where a variety of human activities are allowed (MAB, 2007; UNESCO, 1995).

In the transition zone of biosphere reserves, human activities must clearly be related to environmental improvements, including sustainable use and conservation of biodiversity (Elbakidze et al., 2013; Kratzer, 2018). That is, agricultural and others recreational activities developed in the transition zone are expected to satisfy the first two dimensions of sustainable development including: safeguarding long-term ecological sustainability and satisfying basic human needs (Holden et al., 2014). The rationale for this design is to facilitate a coexistence of human and wildlife thereby implementing a conservation-through use approach which is expected to guarantee the long-term sustainability of these reserve as opposed to old conservation paradigm which advocate for complete protection of protected areas. Understanding how local people use this strategic transition zone and their attitude towards the reserve in that zone is critical for anticipating the ecological impacts of people on biodiversity and the success of these biosphere reserves.

In this study, conducted in one of the largest protected area complex of west Africa, we investigate the sustainability of agricultural practices in the transition zones, the role that roads and patrolling (or lack thereof) by park rangers play in shaping the composition and maintaining biological diversity in agroforestry parklands. We studied the biosphere reserve of Pendjari which is part of the W-Arly-Pendjari (WAP) national parks, a transboundary network of reserves between Niger, Burkina-Faso, and Benin (Amahowe et al., 2013; Harris et al., 2019). Agricultural practices within the transition zone of this biosphere reserve are conducted in traditional agroforestry systems (Djossa et al., 2008; Fifanou et al., 2011). We took advantage of a rare detailed existing dataset (Gaoue, 2000) on the species composition and tree density in traditional agroforestry parks in the transition zone of the Pendjari Biosphere Reserve to assess how the presence of unpaved roads and ethnic diversity affect changes in tree diversity and density in traditional agroforestry systems between 2000 and 2016. We assessed the changes in land-use and land cover over this period to further understand the drivers of changes in biodiversity.

2. Methods

2.1. Study area

The Pendjari Biosphere Reserve covers 4, 661.4 km^2 and is located in the northwest of Benin between 10°30′N - 11°30′N and 0°50 ' - 2°00′E (Fig. 1). The annual rainfall ranges from 1000 to 1100 mm with an average annual temperature of 27 °C. The rainy season last nearly five months from mid-May to October followed by a dry season from November to February (Azihou, 2008). The vegetation is dominated by woodlands, savannas and grasslands established on poorly developed and tropical ferruginous soils (Faure and Volkoff, 1998). Pendjari is organized into three main zones (see Fig. 1): a core area or protected zone (The Pendjari National Park), a buffer zone and a transition zone where sustainable agriculture is permitted (PNP, 2009). The buffer zone is composed of the Pendjari and Konkombri hunting zones. The Pendjari hunting zone is bordered by two main roads (Tanguieta-Batia road, and the Tanguieta-Porga road) which separate it from the transition zone (Assédé et al., 2012). These roads are used by the park rangers to monitor agricultural activities to ensure that they follow sustainable guidelines and to limit illegal poaching and cattle grazing. Along the two main roads and inside the Pendjari hunting zone, local populations were allowed to establish traditional agroforestry systems (Fig. 1) and gather non-timber forest products within the first 5 km perpendicularly from the roads (Houinato and Sinsin, 2000; Téhou et al., 2012). Along these two main roads bordering the Biosphere reserve, two main ethnic groups are established: the Berba ethnic group which accounts for

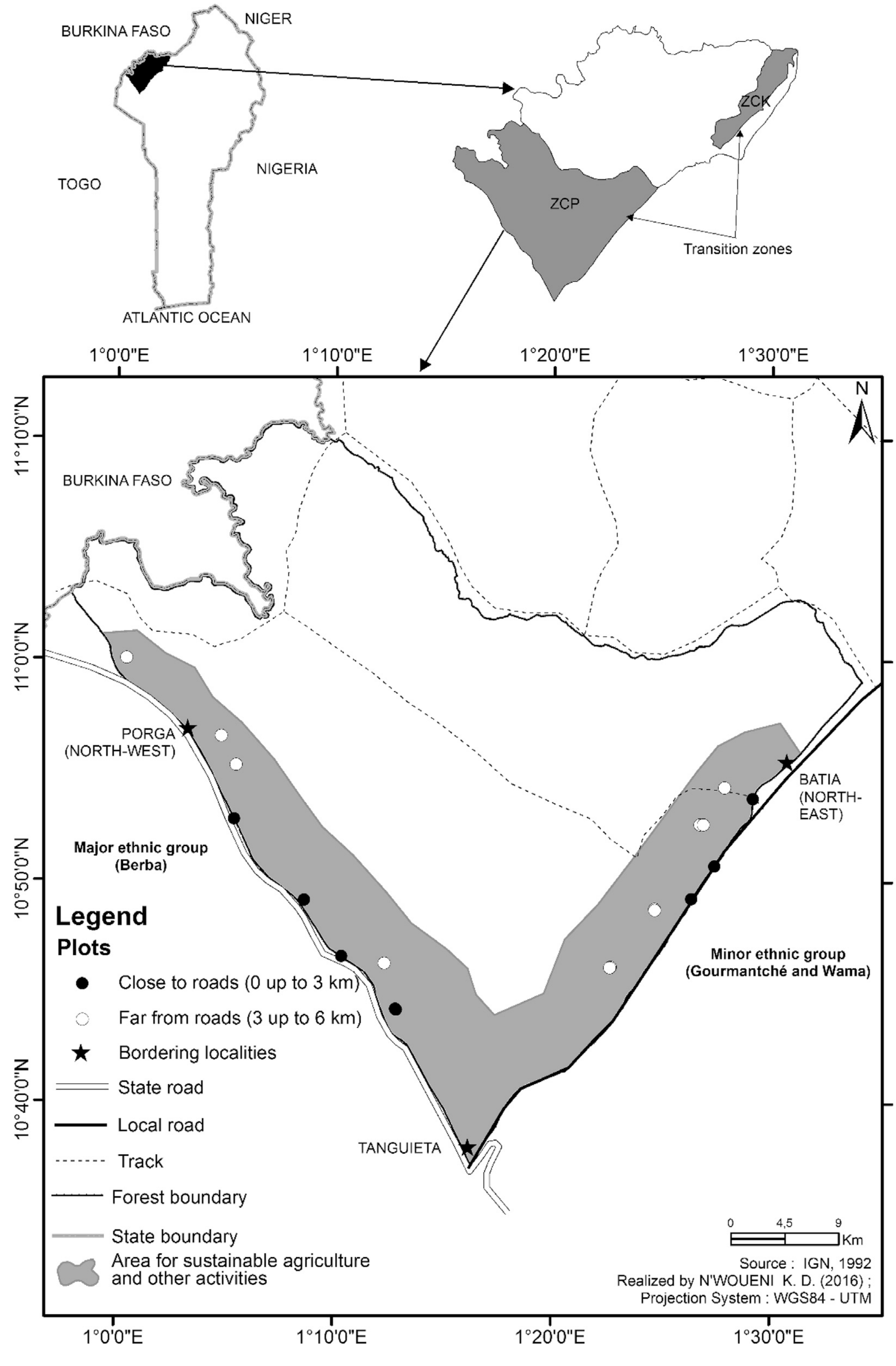

Fig. 1. Distribution of study plots close to or far from the monitoring roads with the major ethnic group on each road within the transition zone of the biosphere reserve of Pendjari. ZCP = Pendjari Hunting Zone, ZCK = Konkombri Hunting Zone.

65% along the northwestern side and Wama and Gourmantché ethnic groups which account for 30% along the northeastern side (Gaoue, 2000). Farms extension in the area occupied by the Wama-Gourmantché is constrained by the Atacora Chain of Mountains.

2.2. Estimating density and diversity

In 2000, we established 20 permanent 50 m × 50 m plots in a factorial design to test the effects of the proximity of agroforestry systems to the main roads ("close" if located < 3 km and "far" if ≥ 3 km), and the ethnicity of the farmers (Berba versus Wama-Gourmantché ethnic groups) on the diversity and density of agroforestry species in the Pendjari (see Gaoue, 2000). We revisited these exact same 20 plots 16 years later in 2016 to measure the diameter at breast height (DBH) of woody individual plants within the plots, their diversity and abundance. Species were identified using local botanical experts and available flora and species lists (Akoegninou et al., 2006; Arbonnier, 2000; Assédé et al., 2012; Gaoue, 2000).

We estimated trees density by counting individual woody plants within each plot. We used this data to develop a matrix of species abundance, each row representing a unique plot and each column representing a species recorded in our survey. We used this matrix to estimate three species diversity metrics: species richness (S), Shannon diversity index (Shannon, 1948) and Pielou evenness (Pielou, 1966). Shannon index and Pielou evenness were estimated at plot level using packages *Hotelling* and *vegan* in R version 3.1.2 (R Core Team, 2015). We estimated the plot level species richness with Chao abundance-based coverage estimators (Chao and Lee, 1992) using the function "*estimateR*" in package *vegan.*

2.3. Statistical analysis

To test the effects of ethnicity and proximity of agroforestry systems to the main roads on tree density, species richness, Shannon index and Pielou evenness we ran a generalized linear mixed effect model (Bolker et al., 2009) in the package lme4 in R, version 3.1.2 (R Core Team, 2015), with ethnicity, proximity and year as fixed effects and plots as random effects. We included year in this model since there was a remarkable variation of density between years. For response variables (species richness and density) for which we detected overdispersion, we used a negative binomial error distribution. For each response variable, we parameterized several candidate models including interactions terms (see Table 1). We selected the best model as the complex model with ΔAICc<2, where ΔAIC is the difference between the Akaike Information Criterion (AIC) of each model and the smallest AIC value obtained; ΔAICc is the small sample corrected version.

To test if variation in trees density was dependent on tree diameter size class distribution, ethnicity and agroforestry systems proximity to the road, we used a log-linear analysis with a negative binomial error to account for overdispersion (Crawley, 2007). We analyzed Landsat-7 ETM and Landsat-8 OLI images (www.earthexplorer.usgs.gov) in Arc GIS version 10.4 to assess change in land cover between 2000 and 2016. Spatial Analyst extension was used for land cover classification and Spatial Statistic was used to estimate the size of land use types.

3. Results

3.1. Effects of roads and ethnicity on tree diversity and density

We found a significant reduction in trees density in the traditional agroforestry systems ($\beta = -1.13$, $Z = -11.08$, $p < 0.001$, Table 2), with an average of 7 trees/ha in 2016 and 20 trees/ha in 2000. This decline in trees density was more remarkable for key traditional agroforestry species such as *Vitellaria paradoxa*, *Parkia biglobosa*, *Ficus gnaphalocarpa* and *Lannea acida*. The average density of *Vitellaria paradoxa* declined from 20 trees/ha in 2000 to 12 trees/ha in 2016. *Parkia biglobosa* and *Ficus gnaphalocarpa* mean densities declined from 16 trees/ha for in 2000 to 4 trees/ha in 2016, and *Lannea acida* density was

Table 1
Parameterized models and selection of appropriate models using information theoretic approach (ΔAICc). P = farms proximity to the road, E = ethnicity of local people around the park, Y = year.

N°	Models	AICc	ΔAICc
1	glmer.nb(Density ~ P + E + Y + P:Y + E:Y + P:E:Y+1|Plots)	300.84	5.68
2	glmer.nb(Density ~ P + E + Y + P:Y + E:Y+1|Plots)	300.84	5.68
3	glmer.nb(Density ~ P + E + Y+1|Plots)	295.16	0.00
4	lmer(Richness ~ P + E + Y + P:Y + E:Y + P:E:Y+1|Plots)	−57.10	116.13
5	lmer(Richness ~ P + E + Y + P:Y + E:Y+1|Plots)	−173.23	0.00
6	lmer(Richness ~ P + E + Y+1|Plots)	−118.79	54.44
7	lmer(Shannon~P + E + Y + P:Y + E:Y + P:E:Y+1|Plots)	48.53	4.82
8	lmer(Shannon~P + E + Y + P:Y + E:Y+1|Plots)	44.40	0.69
9	lmer(Shannon~P + E + Y+1|Plots)	43.71	0.00
10	lmer(Pielou ~ P + E + Y + P:Y + E:Y + P:E:Y+1|Plots)	2.67	2.00
11	lmer(Pielou ~ P + E + Y + P:Y + E:Y+1|Plots)	3.89	3.13
12	lmer(Pielou ~ P + E + Y+1|Plots)	0.76	0.00

Table 2
Effects of ethnicity and proximity of agroforestry systems on trees density, abundance-based coverage estimators for species richness (S_{ACE}), Pielou evenness and Shannon diversity index. β±SE = effect size with standard error.

	Density			**S_{ACE}**			**Shannon diversity index, H**			**Pielou evenness, E**		
	β±SE	*z*	*P*	β±SE	*t*	*P*	β±SE	*t*	*P*	β±SE	*t*	*P*
Intercept	3.43 ± 0.33	10.49	<0.001	0.67 ± 1.97	0.34	0.73	0.26 ± 0.14	1.82	0.06	0.55 ± 0.06	8.92	0.00
Far	−0.80 ± 0.34	−2.34	0.02*	1.22 ± 1.90	0.64	0.52	−0.15 ± 0.15	−1.04	0.29	0.08 ± 0.07	1.14	0.26
Berba	−0.56 ± 0.34	−1.65	0.10	5.87 ± 1.77	3.31	0.00*	0.55 ± 0.15	3.69	0.00*	0.36 ± 0.07	5.04	0.00*
Year 2016	−1.13 ± 0.10	−11.08	<0.001*	0	0	1.00	−0.05 ± 0.17	−0.33	0.74	0.06 ± 0.06	0.98	0.33
Far x Year 2016				0	0	1.00	−0.08 ± 0.17	−0.49	0.63			
Berba x Year 2016				0	0	1.00	−0.46 ± 0.17	−2.70	0.01*			

reduced by half from 16 trees/ha for in 2000 to 8 trees/ha in 2016. However, the density remained constant for species such *Adansonia digitata*, *Combretum nigricans* and *Ficus thoningii*. More critically, species such *Pterocarpus erinaceus*, *Acacia dudgeonii*, *Anogeissus leiocarpa*, *Burkea africana*, *Crossopteryx febrifuga*, *Pericopsis laxiflora*, *Stereospermum kunthianum*, *Tamarindus indica*, *Terminalia avicenioides* and *Vitex doniana* which were present in these agroforestry systems in 2000, disappeared in 2016.

Tree density was significantly lower in agroforests which were far from the main roads than in those close to the main roads ($\beta = -0.80$, $Z = -2.34$, $p = 0.02$), indicating that as farmers establish further from the road, they cut more trees. Tree density were reduced from 19 trees/ha in farms that were close to the main roads and that could be frequently visited by the park guards monitoring team, to 9 trees/ha in farms that were far from these roads. However, there was no significant difference in trees density between ethnic groups ($\beta = -0.56$, $Z = -1.65$, $p = 0.10$). Agroforests in Berba regions had an average of 11 trees/ha while an average of 15 trees/ha was recorded in Gourmantché-Wama farms.

Species richness ($\beta = 5.87$, $t = 3.31$, $p < 0.0001$), Shannon diversity index ($\beta = 0.55$, $t = 3.69$, $p < 0.0001$) and Pielou evenness ($\beta = 0.36$, $t = 5.04$, $p < 0.0001$) were all significantly higher in the Berba farms (7 species, $H' = 0.45$ bit, $E = 0.94$) than in those owned by Gourmantché-Wama (2 species, $H' = 0.09$ bit, $E = 0.56$) but did not vary between 2000 and 2016 regardless of farms positions ($\beta = 1.22$, $t = 0.64$, $p = 0.52$). The effect of ethnicity on Shannon diversity index was year dependent ($\beta = -0.46$, $t = -2.70$, $p = 0.01$) such that in 2016, Shannon index was significantly lower in Berba farms than in those Gourmantché-Wama but this was not true in 2000, even though Shannon diversity index was higher in 2000 (0.45 bit) than in 2016 (0.12 bit). We found no significant difference in Shannon index and Pielou evenness between years and farm positions.

3.2. *Effects of roads and ethnicity on the structure of agroforestry systems*

The size class distribution of woody species in these agroforests was skewed in 2000 (Skewness = 1.65, Fig. 2 a,b) regardless of their position relative to the roads. However, in 2016, the size class distribution was quasi-normal distribution (Skewness = 0.52 to 1.29, Fig. 2 a,b) indicating a significant shift in structure over time. We found a drastic reduction in trees density from 2000 to 2016 particularly in the first three size classes regardless of farms positions: 5–10 cm (in parks far from the roads: $\beta = 2.07$, $Z = 1.96$, $p = 0.04$, Fig. 2a, or close to the roads: $\beta = 3.58$, Z = 3.54, $p < 0.0001$, Figs. 2b), 10–20 cm (in parks far from the roads: $\beta = 3.43$, $Z = 3.38$, $p < 0.0001$ or close to the roads: $\beta = 2.89$, $Z = 2.81$, p < 0.0001) and 20–30 cm (in parks far from the roads: $\beta = 2.89$, $Z = 2.81$, $p = 0.01$ or close to the roads: $\beta = 3.30$, $Z = 3.24$, $p = 0.01$). A similar trend was also observed between years for each ethnic group (Fig. 2 c,d).

3.3. *Change in land cover and land use between 2000 and 2016*

The changes in tree species densities were associated with drastic changes in land use cover in the study region (Fig. 3). Gallery forests cover was drastically reduced by 65.75% from 2530.31 ha in 2016 down from 6447.15 ha in 2000 (Fig. 3a). Gallery forests were converted into cropland (mosaic of field and fallow) and tree savannas, with cropland cover remarkably increased by 20.9% (Figure 3b) to 13182.84 ha observed in 2016 down from 15941.28 ha in 2000 (Fig. 3a). However, an important part of wooded savannas was converted into cropland (Fig. 3c) although tree savannas slightly increased in size by 3.3% from 31481.3 ha in 2000 to 32530.55 ha in 2016. Land covered by houses increased by 95.32% (Fig. 3b) from 5.36 ha in 2000 to 114.54 ha in 2016 (Fig. 3a).

4. Discussion

We investigated the dynamic of traditional agroforestry parks in a West African biosphere reserve. We found a significant decline of trees density between years. The dynamics of traditional agroforestry parks was affected by their proximity to monitoring roads and ethnicity. The drastic decline in trees density observed in these agroforestry parks was surprising for the two indicator species, *Vitellaria paradoxa* and *Parkia biglobosa,* which are expected to be conserved because they provide nuts and seeds with important commercial value to local people (Gnanglè et al., 2012). This controversial trend can be explained by time shifting importance of species harvested for charcoal. *Vitellaria paradoxa* and *Parkia biglobosa* are desirably

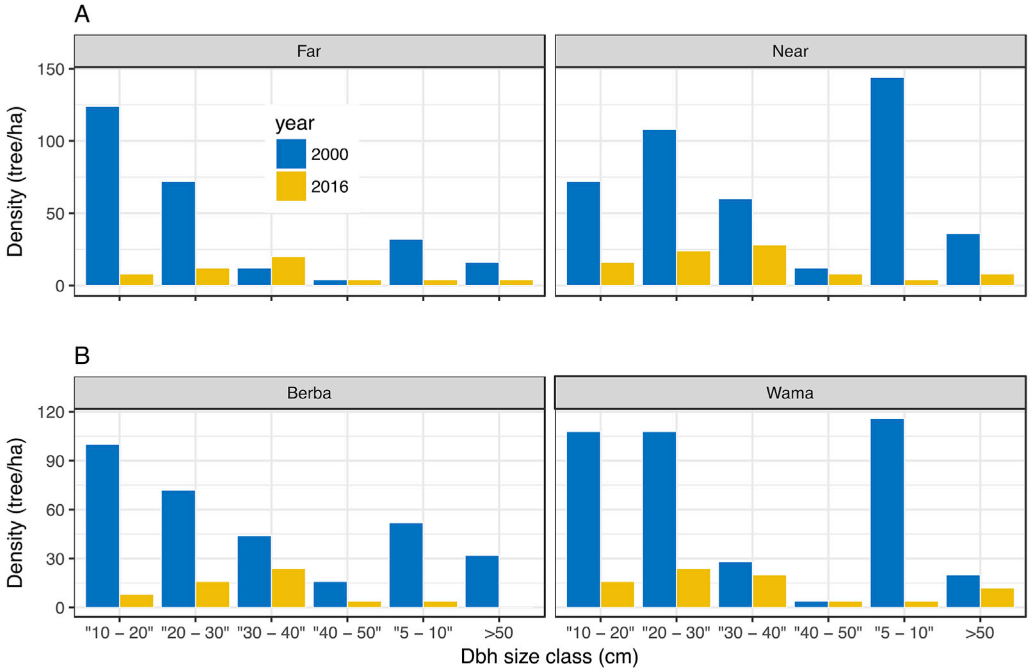

Fig. 2. Effects of farms proximity to main roads and ethnicity on tree diameter size classes distribution: (a) farms close to main roads and (b) farms located far from the main roads in regions dominated by (c) Berba or Gourmantché-Wama ethnic groups between years 2000 and 2016.

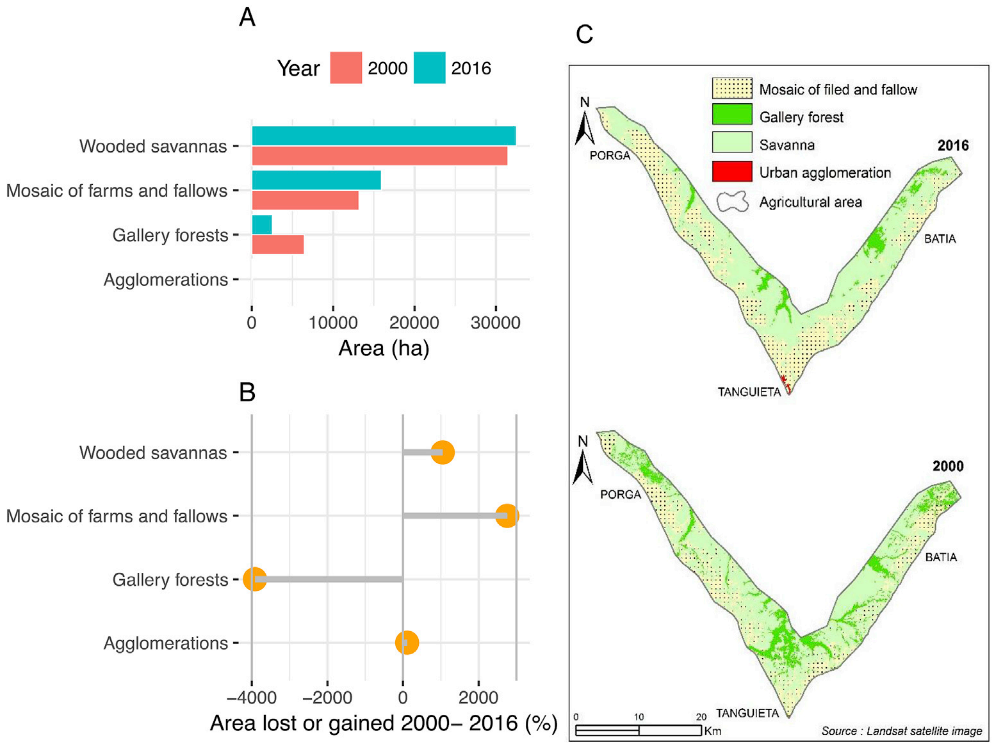

Fig. 3. (a) Differences in land cover classes between 2000 and 2016, (b) variation of land cover losses and gains over time and (c) map showing transitions between land cover classes.

conserved when non-desirable species (e.g., *Pterocarpus erinaceus, Acacia dudgeonii, Burkea africana, Pericopsis laxiflora, Stereospermum kunthianum, Terminalia avicenioides*) are still available in the surrounding vegetation and can meet their demand in firewood. However, when non-desirable species become rare due to overharvesting, local people shift to desirable species, that have long been conserved, as source of firewood. *Vitellaria paradoxa* and *Parkia biglobosa* then become highly sought after for firewood and charcoal (Gaoue, 2000).

We observed a decline in density for trees with DBH <30 cm in traditional agroforestry parks. This result is expected because farmers often conserve large trees which provides non-timber forests products for local communities. Alternatively, trees of small diameter are considered as non-desirable and are harvested for firewood. Even though there is growing evidence that fire and grazing can induce small-size trees (sapling or juvenile) mortality and shift in population structure (Bee et al., 2007; Hoffmann, 1996; Peterson and Reich, 2001), this is not evident in this study since bush fires are not allowed in the biosphere reserve and grazing in agroforestry parklands take place for limited time and in very few places.

The reduction in tree density was also associated with reduction in Shannon diversity index from 2000 to 2016. This is not surprising because some species relative high abundance has decreased in population size and other species have disappeared suggesting that Pielou evenness remained high and constant between years. We found that agroforestry parklands owned by Berba farmers were more diversified with significantly higher species richness, Shannon diversity index and Pielou evenness than traditional agroforestry parks in Gourmantché-Wama region. Trees density was also significantly higher in farms close to the main roads than in those far from the main roads suggesting that farmers attitude and agricultural practices become less sustainable as one move further from the main roads used by park rangers, the "*ecoguardes*", to monitor the transition zone. Farms established far from the roads were monitored less frequently because they were less accessible due to the unfavorable rocky terrain. Several studies emphasized the negative role of roads in protected areas, as they can serve as a gateway for further deforestation, invasion and facilitate transport of illegal hunting products out of the reserves (Fahrig and Rytwinski, 2009; Gelbard and Belnap, 2003; Laurance et al., 2009; McGregor et al., 2008; Trombulak and Frissell, 2000). However, our study provides one positive role play by roads in the facilitation of forest monitoring and biodiversity conservation. The decline in tree density and changes in species composition can also be explained by the weakening of the park monitoring system during the 16 years period. The number of the *ecoguardes* declined steadily, from 21 *ecoguardes* in 2000 to 15 in 2016, and the frequency of agroforestry parks visits also declined over time, from 20 days/month in 2000 to 16 days/month in 2016 (Personal information, Monitoring Director, CENAGREF).

The decline in tree density in agroforestry parks over time was also associated with severe land use changes. In particular, there was a high transition of gallery forests to cropland and savannas from 2000 to 2016. Cropland also gained areas from wooded savanna. This transition of wooded savanna and gallery forests to cropland is consistent with the global trend of world forest dynamic (Achard et al., 2002). Previous studies in West Africa showed a considerable transition of gallery forests to cropland and savannas due to agricultural practices and tree logging for charcoal (Faye et al., 2016; Natta et al., 2002; Ouedraogo et al., 2015). In our study area, 2.83% of annual growth in human population from 2000 to 2013 combined with pronounced soil fertility loss over time (Kombienou et al., 2015) are key drivers of the observed change in land cover. There is a supporting evidence that there is a strong correlation between population growth and density of forest cover. For example, Li et al. (2013) showed that vegetation cover decreased linearly with population density in China. Similar trend was observed in India (Nath and Mwchahary, 2012), in Nigeria (Out et al., 2011) and Ethiopia (Debel et al., 2014). Urbanization also negatively affects forest cover and diversity (Li et al., 2013; Venn et al., 2003). However, urbanization accounted for a very marginal effect in our study because only one traditional agroforestry park was converted into infrastructure during the 16 years period (Fig. 3c). The low transition of cropland to wooded savannas may result from the long period of fallow that conversed into natural vegetation. Similar trend was observed in the community managed forest in South-Eastern Senegal where 3% of cropland was converted into natural vegetation (Faye et al., 2016).

The 16 years regressive trend in the dynamic of traditional agroforestry systems in the current study suggests that agricultural practices in the transition zone of the biosphere reserve of Pendjari are not sustainable and do not guarantee biodiversity conservation. This is consistent with the general trend that African protected areas are not efficient in biodiversity conservation mainly due to human activities, management status, climatic hazards (Pfeifer et al., 2012) and agricultural practices (Tranquilli et al., 2014). For example, a previous study in the transition zone of the same biosphere reserve showed that farming and grazing are not favorable for the wildlife conservation (Houinato and Sinsin, 2000). A similar trend was found in the Oti-Mandouri Wildlife Reserve in Togo showing that agro-pastoral activities are threatening the wildlife (Dimobe et al., 2014). In the Kruger National Park in South Africa, bush fires combined with elephant grazing is the main drivers of drastic decline of diversity and density of large-size trees (Trollope et al., 1998). At a global scale, the negative effect of human activities on forest ecosystems has widely been reported. Although agroforestry is considered as a more reliable management approach to conserve biodiversity (Burgess, 1999; Nair, 2011), this was not the case in the biosphere reserve of Pendjari suggesting that it is critical to revisit the role of traditional agroforestry systems developed by farmers in the conservation of biological diversity in this reserve.

Conflicts of interest

The authors declared no conflict of interests.

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