

## ORIGINAL RESEARCH

# Protection status and national socio-economic context shape land conversion in and around a key transboundary protected area complex in West Africa

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Land conversion, land cover classification, protection status, transboundary protected areas, vegetation loss, WAP

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## Abstract

Transboundary cooperation is being promoted as an effective way to conserve biodiversity that straddles national borders. However, monitoring the ecological outcomes of these large-scale endeavours is challenging, and as a result, the factors and processes likely to shape their effectiveness remain poorly identified and understood. To address this knowledge gap, we tested three hypotheses pertaining to natural vegetation loss across the W-Arly-Pendjari protected area complex, a key biodiversity hotspot in West Africa. Using a new methodology to compare land cover change across large remote areas where independent validation data is unevenly distributed across time, we demonstrate widespread agricultural expansion outside protected areas over the past 13 years. Protected areas with high protection status appear considerably more effective at preventing land conversion than other protected areas. We moreover report marked differences in cropland expansion rates between countries, which we suggest may be linked to differences in rural population growth. Altogether, our results suggest that there can be considerable spatial heterogeneity in anthropogenic pressure across transboundary protected area complexes and call for more comprehensive assessments that capitalize on the current availability of remote sensing information.

## Introduction

States increasingly cooperate across national boundaries to meet global and regional environmental targets. The scientific rationale behind this is that such targets are more likely to be achieved if environmental management occurs at the same scale as the processes that affect environmental outcomes (Hamilton et al. 1996; Petursson et al. 2013). In particular, areas which support a given conservation unit (such as a population or ecosystem) are often shared between multiple countries (López-Hoffman et al. 2010) while many migratory species, which globally sustain key ecosystem functions (Talukdar and Sinha 2013; Bauer and

Hoye 2014), regularly cross national boundaries as part of their life cycle. The transboundary approach, which includes transboundary protected areas (Sandwith et al. 2001) and cooperation for protection of migratory species (Caddell 2005), currently enjoys high-level political support, with four international conventions relevant to biodiversity conservation (namely the Convention on Biological Diversity, the Convention on Migratory Species, the Ramsar Convention and the UNESCO World Heritage Natural Heritage Strategy) explicitly encouraging it.

An important mode of transboundary cooperation is captured by transboundary protected areas (TBPAs). For the purposes of this paper, we define TBPAs as protected

areas spanning “across one or more international boundaries” and involving “some form of cooperation” (IUCN Global Transboundary Conservation Network 2016), rather than just being geographically adjoining protected areas without any management cooperation (Sandwith et al. 2001). The number of TBPAs has been increasing from 59 in the late 1980s (Zbicz 2001) to 227 in 2007 (which represents the most recent assessment; Lysenko et al. 2007), with the International Union for Conservation of Nature (IUCN) identifying the conservation of transboundary ecosystems and migratory species as a key strategy to improve management of protected areas (IUCN 2014). So far, little empirical evidence is available to assess the ecological effectiveness of TBPAs (Busch 2008). Potential benefits of transboundary conservation include protection of larger contiguous areas; more effective responses to threats such as wildfires, poaching and invasive species; and the sharing (and thus the more efficient use) of financial and material resources, information and expertise (Hamilton et al. 1996). However, there are a number of challenges faced by TBPAs that potentially undermine their ability to deliver positive conservation outcomes. Transboundary cooperation requires coordination between countries with different political, economic and/or social contexts and agendas (Perz et al. 2010; Petursson et al. 2013), as well as collaboration between state and non-state stakeholders, including local communities (Duffy 2006). In this context, the lack of empirical insight into the factors shaping the conservation outcomes of such initiatives hampers our ability to improve on the design and implementation of TBPAs, and enhance their cost effectiveness.

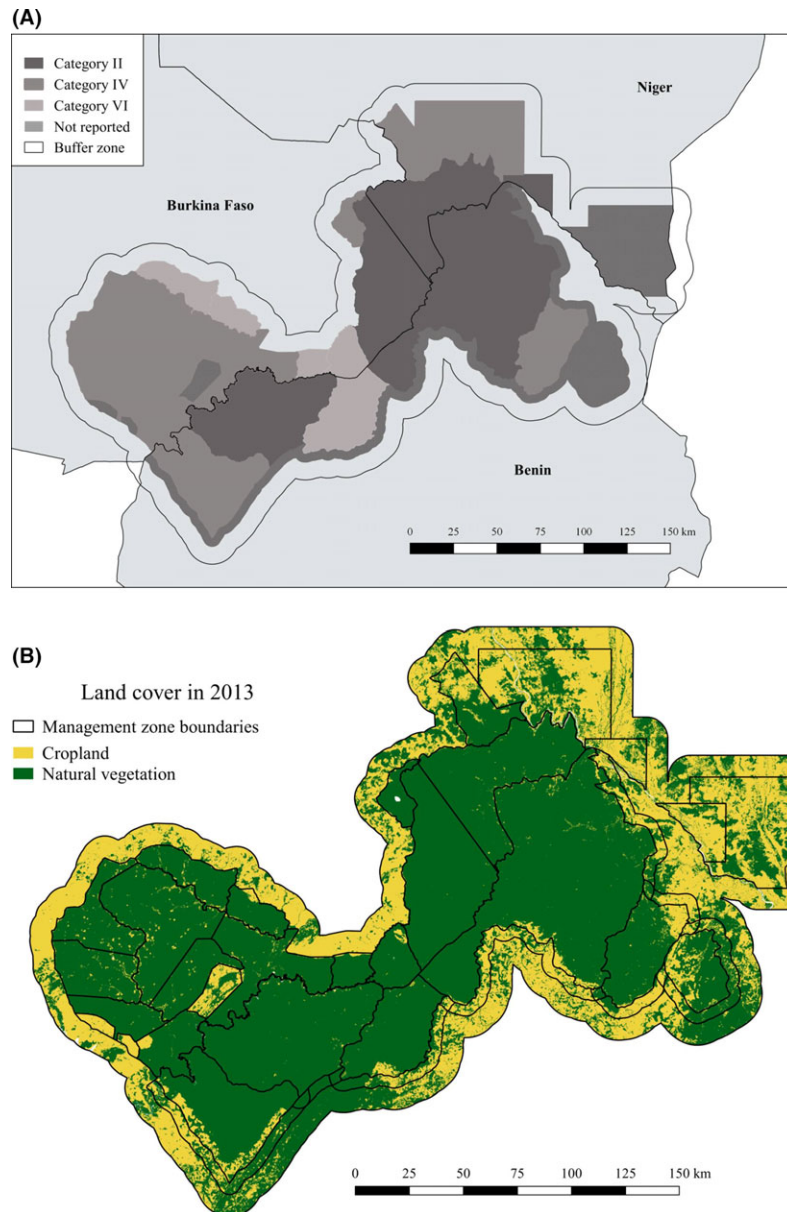
Whilst the economic, social or political impacts of TBPAs have received ample research attention (see e.g. van Amerom 2002; Metcalfe 2003; Duffy 2006; Scovronick and Turpie 2009; Barquet et al. 2014), assessments of ecological outcomes of TBPAs are rare. Taking a national perspective, Reyers (2003) concluded that little additional ecological benefits accrue to South Africa from TBPAs in terms of species diversity, but that diversity of land cover types was increased. At the same time, Plumptre et al. (2007) linked positive developments in Mountain gorilla (*Gorilla beringei beringei*) populations in the Virunga landscape to the cooperation between adjacent PAs across decades of civil war. More recent work suggest that ecological outcomes in a given TBPA could vary significantly between countries (Tang et al. 2010), leading to the recommendation that appraisal of TBPA outcomes must include all participating countries to understand what shapes their overall effectiveness.

Since TBPAs are large and by definition managed by institutions in different countries (Lysenko et al. 2007), carrying out standardized conservation impact assessments

across an entire transboundary landscape is currently still challenging. Satellite remote sensing (SRS) data offers an opportunity to address this problem because it allows the standardized monitoring of multiple protected areas across large spatial extents (Pettorelli et al. 2012; Nagendra et al. 2013). SRS-based approaches have already been used for terrestrial protected area monitoring, to create baseline maps of vegetation, track change in vegetation condition over time and monitor anthropogenic impacts (Gillespie et al. 2008, 2015; Fraser et al. 2009; Huang et al. 2009; Tang et al. 2010). Additionally, using SRS allows simultaneous monitoring of protected areas and their surrounding landscape (Wright et al. 2007; Gillespie et al. 2008), which makes it possible to gauge the impact of pressures arising from outside protected areas (DeFries et al. 2010; Laurance et al. 2012). Because time series of SRS data often span decades (Kuenzer et al. 2014), change in conservation outcomes of TBPAs can moreover be tracked.

Here, we illustrate this opportunity using open-source satellite and high resolution optical imagery to investigate the impact of anthropogenic pressure from land cover change in and around the W-Arly-Pendjari (WAP) complex. The WAP complex is a large TBPA in West Africa, comprising protected areas in Benin, Burkina Faso and Niger (Fig. 1A). To evaluate the ecological outcome of this TBPA, we decided to assess the ability of the WAP complex to reduce pressures on biodiversity arising from agricultural expansion. We used Landsat imagery to classify land cover in and around the WAP, since it adequately captures landscape patterns (Townsend et al. 2009) and is commonly used for land cover classifications in and around protected areas (e.g. Fraser et al. 2009; Huang et al. 2009; Tang et al. 2010). While doing so, we tested three hypotheses (summarized in Table 1).

- Hypothesis: 1 Protected areas should be more effective at protecting natural vegetation from agricultural expansion than the surrounding buffer zone (Bruner et al. 2001; DeFries et al. 2005; Andam et al. 2008), meaning that agricultural encroachment would be faster in the buffer zone than in protected areas.
- Hypothesis: 2 Protected areas with a higher level of protection should be more effective at protecting natural vegetation than protected areas with a lower protection status (Joppa et al. 2008). In particular, we expected protected areas of IUCN category I and II to show lower levels of agricultural encroachment than other protected area types found in the WAP complex.



**Figure 1.** (A) Overview over the study site. Boundaries of protected areas and buffer zones (Grange, 2016, personal comm.) except for those in Benin, which were taken from WDPA (WDPA, 2016). (B) Land cover in protected areas and buffer zones of the WAP in 2013.

**Hypothesis: 3** A host of socio-economic and demographic drivers have been invoked to explain agricultural expansion, including macroeconomic context, agricultural policies and economic returns on agricultural activity, that may interact to shape land use change (Mertens and Lambin 2000; Reid et al. 2000; Lambin et al. 2001; Umemiya et al. 2010; Ferretti-Gallon and Busch 2014). However, empirical studies have shown

that agricultural expansion around the WAP strongly correlates with population pressure (Konrad 2006; Ouedraogo 2006, 2010; Houessou et al. 2013). Therefore, we hypothesized that agricultural expansion would be highest in Niger, which experienced the highest rural population growth between 2000 and 2013, followed by Burkina Faso and Benin (World Development Indicators, The World Bank 2016).

**Table 1.** Overview of hypotheses.

	Name	Description	Prediction
Hypothesis 1	Effectiveness of PAs	PAs are better at protecting habitat from anthropogenic pressures than surrounding buffer zones (Bruner et al. 2001; DeFries et al. 2005; Andam et al. 2008).	Loss of natural vegetation due to cropland expansion is slower inside PAs than outside.
Hypothesis 2	Differences in anthropogenic pressures between PAs of different legal status.	PAs with higher legal protection status are better at protecting habitat from anthropogenic pressures than PAs with lower status (Joppa et al. 2008).	PAs of IUCN category II <sup>1</sup> have lower natural vegetation loss rates than PAs with lower or no reported IUCN status.
Hypothesis 3	Differences in anthropogenic pressures between countries	Agricultural expansion is higher in countries with faster rural population growth (Konrad 2006; Ouedraogo 2006, 2010; Houessou et al. 2013).	Cropland expansion is faster in Niger than Benin and Burkina Faso.

PAs here stands for protected areas.

<sup>1</sup>Highest IUCN category of PAs in the WAP complex.

## Materials and Methods

### Study area

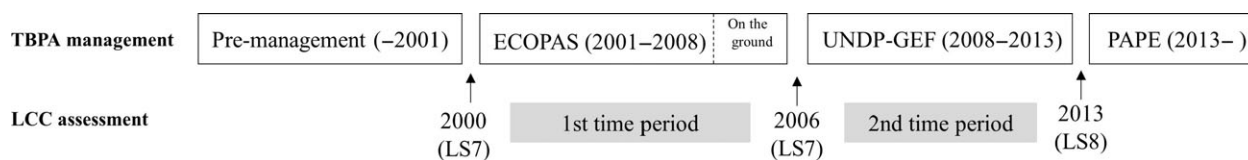
Described as the “largest and most important continuum of terrestrial, semi-aquatic and aquatic ecosystems in the West African savannah belt” (Amahowé et al. 2013), the WAP complex is an important area for biodiversity, holding e.g. more than half of West African lions (Henschel et al. 2014). Apart from its significance for biodiversity, the WAP is also an example for active cooperation between several countries in the management of a TBPA, with formal cooperation between Benin, Burkina Faso and Niger starting in 2001 (Programme Régional Parc W/ECOPAS 2005; Accord relative à la gestion concertée de la Réserve de Biosphère Transfrontalière du W 2008). On-the-ground management started in 2006 as part of the EU-funded ECOPAS project (Amahowé et al. 2013) and has since been continued during successive projects (Fig. 2).

The WAP complex comprises core protected areas and associated buffer zones, where agriculture is allowed (Accord relative à la gestion concertée de la Réserve de Biosphère Transfrontalière du W 2008; Fig. 1A). There are four protected areas of IUCN category II in the WAP (Parc W de Niger, Parc W de Burkina Faso, Parc W de Benin and Boucle de la Pendjari in Benin). There are also protected areas of category IV and VI, as well as protected areas for which no information on categorization is available (Fig. 1A). The boundaries of the buffer zones (“zones de transition”) were defined as a 10 km buffer around all core zones for the purposes of this study, since no geo-referenced shapefile capturing official buffer zone

boundaries could be sourced. This distance was intended to be a conservative assumption about the extent of the buffer zones, and capture land cover dynamics immediately adjacent to protected area boundaries. The WAP buffer zone has a size of c. 14,000 km<sup>2</sup>, whilst the protected areas have a size of c. 36,000 km<sup>2</sup>.

Climatic conditions across the WAP complex range from dry, Sahelian climate in the North (c. 500 mm annual precipitation) to wetter, Sudanese-Guinean climate in the South (c. 1200 mm annual precipitation; UNDP 2007). Towards the coasts, the dry season is punctuated by two rainy seasons lasting from April-June and September-October, whereas more Northern areas experience a single wet season from July-September (WMO 2013). Across the study area, elevation ranges from 130–630 m (SRTM DEM 30 m, data courtesy of U.S. Geological Survey). The WAP complex is dominated by grassland, shrub savannah and savannah woodlands, with some gallery forests and riparian vegetation/marshlands around water bodies (Clerici et al. 2007).

Approximately one million people live within 40 km of formally protected areas of the WAP complex (UNDP-GEF 2004). Since the mid-1990s, cotton cultivation has been replacing subsistence agriculture as the main economic activity of the area (UNDP-GEF 2004), especially around riparian zones in Benin and Burkina Faso (Amahowé et al. 2013). Cropland extent has increased around protected areas in parts of Benin and Burkina Faso in recent decades (Ouedraogo 2006, 2009, 2010; Houessou et al. 2013) and agricultural encroachment is expected to be a primary threat to biodiversity (UNDP 2014). The extent of cropland outside of protected areas around the



**Figure 2.** WAP management periods and year of acquisition of Landsat imagery used for land cover classification (Amahowé et al. 2013; Programme Régional Parc W/ECOPAS 2005; Ndiaye 2014; UNDP 2014).

WAP is believed to have been increasing in Benin (Houessou et al. 2013) and Burkina Faso (Ouedraogo 2006, 2009, 2010), while areas are known to have been cleared for agriculture inside formally protected areas in Benin (Houinato and Sinsin 2000). In addition, pastoralists are increasingly compressed into smaller areas as a result of land shortage, and this has led to land degradation inside the protected areas (UNDP-GEF 2004), even though many on-the-ground conservation activities in the WAP have been focused on limiting and controlling the movement of livestock through the complex (Amahowé et al. 2013).

## Data

Landsat Surface Reflectance products were obtained for Landsat 8 (OLI) for the year 2013 and Landsat 7 (ETM+) for the years 2000 and 2006 (data courtesy of U.S. Geological Survey; see Table S1; Masek et al. 2006; Vermote et al. 2016). Landsat 7 ETM+ and Landsat 8 OLI Surface Reflectance products are corrected for atmospheric effects by the LEDAPS and L8SR algorithm respectively, which have been validated elsewhere (e.g. Claverie et al. 2015; Vermote et al. 2016). We assume that bands 2–7 from Landsat 8 correspond to bands 1–5 and 7 in Landsat 7, since the surface reflectance values derived from these two sensors differ by only *c.* 2% (Flood 2014; Mishra et al. 2014).

Years for analysis were chosen to coincide with changes in management periods as closely as possible (Fig. 2). For each year, scenes from April–October were chosen to represent the wet season (scenes towards the end of the wet season were preferred), and scenes from December–February were chosen to represent the dry season, to allow land-cover specific seasonal changes to be represented. Scenes with <5% cloud cover were preferred (as described in the metadata associated with Landsat scenes), as were adjacent scenes acquired on the same date (to facilitate seamless merging). In 2003, the Scan Line Corrector of Landsat 7 failed, so no surface reflectance information was available for *c.* a third of the study area in 2006 (Markham et al. 2004).

All pre-processing was carried out in R (version 3.2.5, R Core Team 2016), using the packages *raster* (Hijmans

2016) and *RStoolbox* (Leutner and Horning 2016). Pixels covered by clouds and water were identified from the relevant masks included in the Surface Reflectance products and excluded from subsequent analysis. Scenes were histogram matched where necessary (Richards 2006). In these cases, the central scenes, which covered large parts of the study area (path 192 and 193), were chosen as master images, while adjacent scenes that covered smaller parts were used as slave image. After histogram matching, scenes were merged and cropped to the extent of the study area.

## Land cover classification and validation

For each year (2000, 2006 and 2013), a supervised land cover classification was used to distinguish between cropland and natural vegetation using the Random Forest classifier with five-fold cross validation, three tuning parameter levels and using 20,000 pixels to train the model (Breiman 2001; Kuhn 2008). For the year 2013, training pixels were identified from high-resolution Google Earth imagery (Google Earth version 7.1.5.1557, May 20, 2015). We distinguished two broad land cover classes: cropland, and natural vegetation, which includes grassland, shrub savannah, savannah woodlands, and gallery forests and riparian vegetation/marshlands. Training pixels were randomly split into training and validation pixels before classification. In total, 40% of original pixels were used for validation. The supervised classification algorithm used a combination of Landsat surface reflectance, tasseled cap-transformed bands and texture metrics from both the dry and the wet period ( $n = 48$ , see Table S2; Huang et al. 2002; Baig et al. 2014), since tasseled cap dimensions and textures provide complementary information about the spatial distribution of reflectance values and help distinguish between land cover classes (Chen et al. 2004; Lloyd et al. 2004).

For 2000 and 2006, image availability on Google Earth was too small to provide both training and validation data. To obtain training data for these years, we filtered the 2013 training data, retaining only those pixels which had not undergone land cover change. We assumed that pixels which have undergone land cover change show larger changes in measured reflectance than pixels which have not.

Change in measured reflectance across all bands was quantified as follows. The first two principal components (PCs) of all layers used in the 2013 land cover classification (Table S2) were calculated using a standardized PCA. Since bands from both the dry and the wet season were used for the PCA, characteristic changes in phenology across the year are captured by the PC scores. PC1 and PC2 explained 70.5% of variability between pixels, meaning that they captured the majority of spectral variability across the WAP complex. For 2000 and 2006, the same layers were generated. Following standardization of the layers, the loadings of the 2013 layers on PC1 and PC2 were used to calculate “pseudo-”PC scores for 2000 and 2006, respectively. Spectral change vector analysis was then used to quantify the magnitude of change each pixel had undergone in the two-dimensional space of PC1 and PC2.

Based on this analysis, training pixels for 2000 and 2006 were selected from the 2013 training data. Specifically, all pixels showing an above-median change in measured reflectance were excluded from the training data. This was based on the observation that land cover transitions between cropland and natural vegetation had been shown to be rarer than stable land cover in the WAP during the preceding decades (Clerici et al. 2007). This means that the change in measured reflectance observed for the majority of pixels will be due to influences other than land cover change (e.g. differences in the SRS sensors, or phenology). Thus, above-median change in measured reflectance will reflect an exceptional amount of change, indicating that land cover change has occurred.

A supervised classification was then run for each year using the identified training pixels and the same model parameters as for 2013. Again, land cover maps were internally validated by splitting training pixels into training and validation pixels before model training. External validation was carried out with validation pixels derived from Google Earth imagery for 2006. No independent validation (i.e., using validation data from Google Earth imagery from the same year) could be carried out for 2000 since there was no high-resolution Google Earth imagery available for this time. Instead, the 2000 land cover map was validated with the “no change” pixels identified by the spectral change analysis.

## Land cover change analysis

The land cover maps from 2000, 2006 and 2013 were used to quantify changes in cropland extent between 2000 and 2006 (first time period) and 2006 and 2013 (second time period, Fig. 2) in both protected areas and buffer zones. For the purpose of land cover change analysis, any pixels which had missing data for one or both years of a given time period were excluded (since land cover change

cannot be detected if land cover information is not available for both points in time). We calculated the loss of natural vegetation as the proportion of natural vegetation at time  $t$  which had been converted to cropland by time  $t + 1$ . This metric shows what proportion of natural vegetation was affected by conversion to cropland in a given time period.

## Results

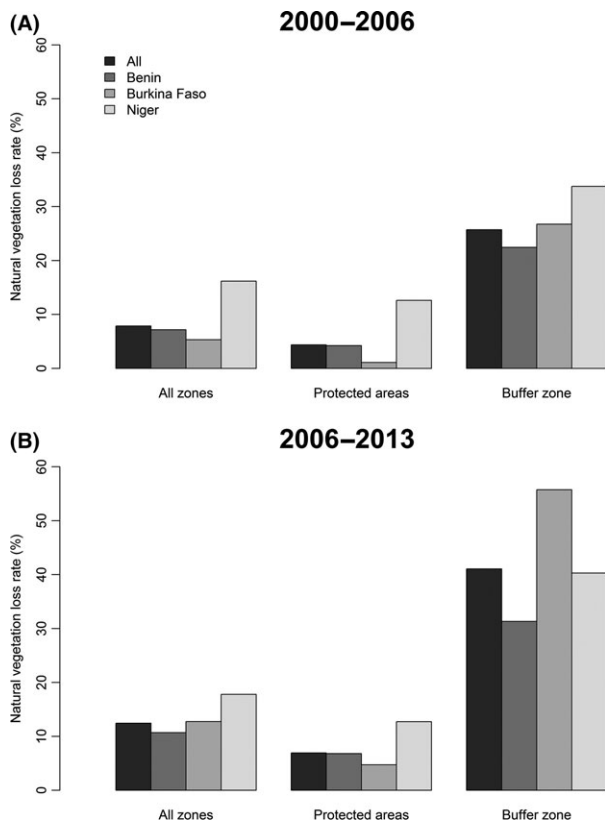
Land cover classification was highly accurate in all 3 years (Table 2). Overall accuracy of the land cover classification was 84%, 91% and 88% for 2000, 2006 and 2013 respectively (Table 2). User and producer accuracy for both land cover classes were high in all 3 years (Table 2). In total, 5.1%, 30.6% and 0.3% of the study area could not be classified in 2000, 2006 and 2013, respectively. In 2013, cropland covered 17,850 km<sup>2</sup> (35.7%) of the WAP, the majority of which (56.5%) occurred in the buffer zone (Fig. 1B). However, cropland was also present in protected areas in all three countries. This general pattern held for 2000 and 2006 (see Fig. S1). The extent of cropland was lower in 2000 (15,200 km<sup>2</sup> or 30.4% of the WAP) and in 2006 (14,500 km<sup>2</sup> or 29% of the WAP).

As expected under Hypothesis 1, protected areas were more effective than surrounding buffer zones at protecting natural vegetation (Fig. 3). Between 2000 and 2013, protected areas collectively lost 6.4% of their natural vegetation (c. 2280 km<sup>2</sup>), whereas buffer zones lost 24.4% (c. 3410 km<sup>2</sup>). If natural vegetation was lost at an equal rate each year, this corresponds to annual loss rates of 0.5% and 2.1%, respectively.

As expected from Hypothesis 2, also, protected areas with higher level of protection were better at preventing agricultural encroachment (Fig. 4). In protected areas of IUCN category II, ~1% of natural vegetation was converted to agriculture in both time periods. A similar rate of ~1% was observed for protected areas of category VI between 2000 and 2006, but land conversion during this period was higher in protected areas of category IV (4.9%; Fig. 4). Between 2006 and 2013, the rate of natural

**Table 2.** Overall accuracy of the land cover classifications, as well as user and producer accuracy for cropland and natural vegetation.

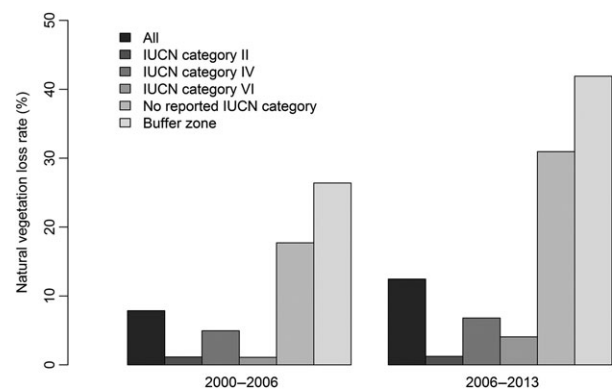
Year	Overall accuracy (%)	User accuracy		Producer accuracy	
		Cropland (%)	Natural vegetation (%)	Cropland (%)	Natural vegetation (%)
2000	84	77.8	88.7	75.8	87.6
2006	91	95.3	87.6	85.5	96.1
2013	88	85.2	98.3	91.2	98.1



**Figure 3.** Between-country differences in loss of natural vegetation, that is, the proportion of remaining natural vegetation that was converted to cropland.

vegetation conversion however markedly increased in these protected areas, with 6.8% and 4.1% of natural vegetation converted in protected areas of category IV and VI respectively over this period, compared to the 2000–2006 period. Interestingly, even more natural vegetation was lost from protected areas with no reported category (17.7% and 30.9% between 2000–2006 and 2006–2013, respectively). However, these rates were still lower than those observed in the buffer zones, where 26.4% and then 41.9% of natural vegetation was lost during the two successive time periods.

As expected from Hypothesis 3, finally, agricultural expansion rate differed between countries. Between 2000 and 2006, cropland expansion was substantially higher in Niger (in both protected areas and the buffer zone; 16.2%) than in Benin (7.2%) and Burkina Faso (5.3%; Fig. 3), which was expected given differences in rural population growth rates. Between 2006 and 2013, however, the cropland expansion in the buffer zone was highest in Burkina Faso (55.7%), followed by Niger (40.3%) and Benin (31.3%), which was unexpected given the difference in rural population growth. In protected areas, however, cropland expansion was highest in Niger, as expected.



**Figure 4.** Differences in loss of natural vegetation, that is, the proportion of remaining natural vegetation that was converted to cropland, between protected areas of IUCN category II, IV, VI, those without an IUCN category, and buffer zones.

## Discussion

To the authors' knowledge, this is the first study that quantifies the ecological outcomes of an entire TBPA over more than a decade of transboundary cooperation, adding an important dimension to previous assessments, which have focussed on quantifying management activities (Zbicz 2001; Schindler et al. 2011; Büscher 2012; Amahowé et al. 2013) and investigating TBPA effects on local communities (Wolmer 2003) or political structures (Duffy 2006; Büscher 2012), often using qualitative data. Admittedly, our analysis was limited by data availability, notably ground-truthed land cover information across the time period considered here (2000–2013). Instead, open access, very high resolution optical imagery was used to distinguish between different type of land cover. This type of imagery was not available across the entirety of the WAP in 2000 and 2006, so we had to generate training data from spectral change analysis for these years. There was however enough very high resolution optical imagery to validate the 2006 land cover map with independent data from the same year, and the accuracy of this map was high. We are therefore confident that our approach produced reliable land cover maps, allowing the monitoring of land cover change across more than a decade. Additionally, failure of the Scan Line Corrector of Landsat 7 in 2003 meant that there was no spectral information for a third of the study area in 2006 (Markham et al. 2004). However, data loss occurred along parallel stripes, meaning that it was equally distributed in space (both between countries and between the buffer zone and protected areas; Fig. S1). Therefore, the remaining data were very likely a balanced sample of land cover across the WAP.

Based on our results, it is clear that the WAP complex has been under increasing pressure from expansion of cropland over the past decade. The coalition of protected

areas has been successful at protecting natural vegetation from agricultural expansion, and this result echoes previous observations in other West and Central African protected areas (Struhsaker et al. 2005) as well as global protected area assessment reports (DeFries et al. 2005; Gaston et al. 2008). An earlier assessment of land cover change around the WAP (Clerici et al. 2007) found that protected area boundaries prevented cropland expansion between 1984 and 2002, meaning that they have successfully buffered habitats from anthropogenic pressures in an area of high population growth for at least three decades.

Interestingly, the ability to buffer habitats from anthropogenic pressure varied considerably between protected areas. Specifically, protected areas of IUCN category II lost less natural vegetation than protected areas with lower or no reported status. This supports previous observations that little or no habitat has been lost in protected areas of categories I and II in tropical areas (DeFries et al. 2005). At this stage, we can only hypothesize that category II protected areas in the WAP may be more effective because they are better enforced or funded than other protected areas, or because their boundaries are protected by adjacent protected areas (Fig. 1A; Bruner et al. 2001). Surprisingly, protected areas of category VI retained a higher proportion of natural vegetation than category IV, suggesting that sustainable use of natural resources is compatible with protection of natural vegetation in the WAP.

Our results also showed that loss of natural vegetation occurred at markedly different rates in different countries. Surprisingly, Burkina Faso had the highest cropland expansion rates in buffer zones between 2006 and 2013, even though it did not have the highest rate of rural population growth (World Development Indicators, The World Bank 2016). Understanding these between-country differences in cropland expansion requires identifying the drivers of land use change around the WAP. For instance, in the South of Burkina Faso, increases in per capita cropland in Eastern Burkina Faso have been reported between 2001 and 2014 (Knauer et al. 2017). More often, cropland expansion has been linked to population growth driven by within-country migration (Ouedraogo et al. 2009; Paré et al. 2014). Conditions for rainfed agriculture are better in south Burkina Faso, which falls in the Sudanian climate zone, than in the North, which is located in the drier Sahelian zone (Ouedraogo et al. 2006), prompting farmers to migrate to the South in search of cultivable land. However, there is even scarcer information about ultimate drivers of land use change in the Benin and Niger part of the WAP. Our results thus illustrate that rural population growth rates do not fully capture the relationships between human activity and cropland expansion, and call for more research to identify the other possible factors driving crop expansion rates in the WAP, to inform future conservation actions.

Interestingly, conversion rates after 2006 were generally larger than before, reflecting increasing pressure on the WAP. It is conceivable that faster cropland expansion after 2006 occurred partly as a response to increased food insecurity after the food price crisis in 2007/08 (Mertens and Lambin 2000), in the context of quickly increasing populations (UN-ESA, Population Division 2015). If this is the case, then maintaining the integrity of the protected areas of the WAP in the future will likely become even more challenging, since population growth and climate change are predicted to increase the demand for scarce cultivable land even further (Niasse et al. 2004; Roudier et al. 2011). In this context, conservation actions aimed at alleviating the demand for land around the WAP in the long-term, either by increasing agricultural output or by developing alternative sources of livelihoods, might be recommended. Despite the problems faced by the WAP, it is remarkable that this protected area complex still encompasses the majority of West Africa's lions and elephants (IUCN 2007; Henschel et al. 2014), as well as the only remaining population of cheetah for the region (IUCN SSC 2012). By this measure, it is, therefore, doing better than most other protected areas in West Africa. However, our analysis shows that its future is by no means secure, and it is important that governments, conservationists and scientists work together to safeguard this, the most important west African landscape for large mammal conservation.

Altogether, our results should be interpreted as a conservative estimate of ecological conservation outcomes in the WAP. We did not distinguish between different types of savannah, and thus were not able to quantify habitat degradation or loss of particular habitat types, for instance due to transhumance (Bouche et al. 2004; UNDP-GEF 2004). Hence, further research should focus on translating the natural vegetation losses we report into changes in habitat availability and quality for key species by taking into account habitat diversity, fragmentation and degradation. This would allow quantifying the effect of land cover change on biodiversity in the WAP. Many populations of large mammals in the WAP have been decreasing since the 1980s (Konrad 2006), and the loss of natural vegetation and increasing isolation of the WAP from the wider landscape since 2000 may have reduced its carrying capacity, which could decrease wildlife populations further (Fahrig 2003; Clerici et al. 2007).

Ultimately, our work suggests that transboundary conservation initiatives do not represent an easy fix to the conservation challenges we face. TBPA's must be backed up by significant political will and resources if they are to deliver conservation outcomes. In order to inform such efforts, it is clear that more attention must be given to the monitoring of their ecological outcomes. We here



demonstrate how capitalizing on currently available satellite imagery allows addressing this information gap. Pairing open source remote sensing with aerial imagery, spectral change analyses and supervised classification algorithms could be a cost-effective way to quantify long-term land cover change in and around TBPA for which no ground-truthed land cover data is available. Crucially, our methodology can easily be repeated over time, allowing close monitoring of TBPA outcomes, even where independent validation data is unevenly distributed across time. This would allow gauging where land cover change is likely to pose a significant threat to a TBPA, and could then guide subsequent on-the-ground assessments of habitat types and species distribution, allowing successful cases of transboundary cooperation for conservation to be identified in order to develop best-practise cooperation guidelines. Eventually, this information could inform the design of future transboundary conservation interventions, allowing prioritization of areas and choosing appropriate interventions (Margules and Pressey 2000).

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We thank Philipp Henschel for providing information on IUCN categories of WAP protected areas. Landsat Surface Reflectance products courtesy of the U.S. Geological Survey.

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## Supporting Information

Additional supporting information may be found online in the supporting information tab for this article.

**Table S1.** Landsat Surface Reflectance images used for land cover classification. Wet season images were acquired between April and October of a given year, dry season images between December and February (of the following year).

**Table S2.** Band combination used for land cover classification.

**Figure S1.** Land cover maps of 2000 and 2006. Diffuse data gaps in Landsat 7 imagery for 2006 had were due to failure of the Scan Line Corrector (Markham et al. 2004). A total of 30.6% of pixels could not be classified because of this (27%, 33% and 38% of pixels were empty in Niger, Benin and Burkina Faso, respectively; 31% were empty in protected areas and 34% in the buffer zone). Spatially concentrated, “blotchy” data gaps were due to water or cloud presence, which were masked out using the cloud and water mask provided with Landsat Surface Reflectance products (see Methods).