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Resilience, remoteness and war shape the land cover dynamics in one of the world's largest miombo woodlands☆

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ABSTRACT

The highlands of southeast Angola are one of the world's largest intact formations of miombo woodland. Recent interest from conservation groups is increasing the possibility of a new protected area in this conflict-afflicted, remote region, contributing to the "30 × 30" target of the Global Biodiversity Framework. With the potential for a new protected area, it is important to quantify the extent and change of natural and anthropogenic land covers in the region, not least because of the close dependence of livelihoods on natural resources in the miombo. We developed a 1990–2020 land cover time series, analysing deforestation, canopy opening, canopy closure, and vegetation regrowth after disturbance. Regional woodland extent has remained roughly constant despite frequent transitions between dense and open woodlands. Canopy opening peaked post-civil war, potentially related to the resettlement of displaced people. Over 30 years, 61 % ± 2 % of canopy opening was offset by subsequent canopy closure, which peaked a decade after the war ended, indicating the resilience of miombo systems. A woodland resource-use frontier, consisting of deforestation and canopy opening, is evident in the north-west of the area, likely driven by urban demand for agricultural products, charcoal, timber and other wood-derived goods. A distinct "core" of dynamic woodland occupies 52 % of the study region, where there is no evidence that shifting cultivation and local livelihoods are a net cause of land cover change. We do not find evidence for extensive net woody encroachment, only 2 % of the study region is being encroached by woody vegetation. This canopy closure is associated with remoteness from anthropogenic pressures and biophysical drivers that facilitate woody vegetation growth. Policymakers and conservation managers can use these data to aid in locating and prioritising interventions to sustainably produce agricultural and wood fuel products to meet increasing urban demand. Additionally, supporting conditions for maintaining both biophysical processes and livelihoods in remote areas is crucial to achieving 30 × 30 equitably.

Introduction

The miombo woodlands of southern Africa harbour unique biodiversity and are a major terrestrial carbon store, in addition to providing a diverse array of provisioning ecosystem services (McNicol et al., 2018; Ribeiro et al., 2020; Ryan et al., 2016). In southern Africa, the miombo woodlands cover up to 1969,000 km^2 and are found in the southeastern Democratic Republic of Congo, Angola, Zambia, western and southern Tanzania, central and northern Mozambique, northern Zimbabwe and Malawi

(Ribeiro et al., 2020; White, 1983). Small and large-scale agriculture, charcoal production and timber harvesting are recognised as the leading activities causing biomass loss in the miombo region (Dziba et al., 2020). Cropland areas for seven miombo countries increased from 100,000 to 272,000 km^2 from 1961 to 2014, whilst rural population increased from 31 to 111 million from 1961 to 2020 (FAOSTAT, 2023). This increasing trend of conversion of woodland to cropland and pressure on resources is set to continue (UNCEA, 2011). Development of agricultural frontiers

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and rapid urbanisation increases the likelihood of deforestation and fragmentation of contiguous areas of woodland (Buchadas et al., 2022; IPBES, 2018). Presently, 5.4 % of miombo woodlands are under IUCN category Ia, Ib or II protected areas, with 3.6 % of Angolan miombo under these categories (Dziba et al., 2020). For conservationists to realise the Convention on Biological Diversity's (CBD) ambitious vision for 30 % area-based conservation coverage by 2030, agreed upon at the COP15 summit (Gilbert, 2022), an appropriate approach may be to protect 30 % of each ecoregion within Africa. This ecoregion-based approach presents a pragmatic tool for achieving area-based conservation targets in an ecologically representative manner (Dinerstein et al., 2017; Magnusson and Gering, 2004), whilst 30 % land conservation is widely being proposed by the conservation community as an aspirational milestone to protecting half of the planet by 2050 and may halve the extinction risk of terrestrial biodiversity (Dinerstein et al., 2019; Hannah et al., 2020). Therefore, a target of protecting 30 % of the miombo ecoregion might be considered.

Protected areas and area-based conservation remain prominent strategies for biodiversity conservation and avoiding carbon emissions through deforestation and degradation (Buchadas et al., 2023; Duncanson et al., 2023; McNicol et al., 2023). The sparsely populated and mostly inaccessible woodlands of the highlands of southeast Angola are the largest somewhat contiguous area of woodland in Angola and represent one of the largest contiguous blocks of miombo woodland in the world. Despite not being identified on global conservation priority maps (Jung et al., 2021), the remoteness and intactness of the highlands of south-east Angola make the area a prime candidate for area-based conservation measures for conservationists to achieve 30 × 30 in the miombo ecoregion.

The Angolan Highlands, known by some local people as "Lisima lya Mwono", "the source of life", encompass the headwaters of the Okavango, Zambezi and Kwando rivers. Lisima is hydrologically important, supporting peatland deposits and regulating seasonal flows of water that support the livelihoods and the well-being of people across Angola, Namibia, Botswana, Zambia, Zimbabwe and Mozambique, as well as biodiversity in the Okavango delta (Lourenco et al., 2022a; Pröpper et al., 2015).

Recent literature investigating a range of land system and ecosystem science questions in south-east Angola, and current efforts by the National Geographic Okavango Wilderness Project (NGOWP) have raised the profile of the Angolan Highlands and the surrounding areas (Gomes et al., 2021; Gonçalves et al., 2017; Loft et al., 2024; Lourenco et al., 2022a, 2022b; Mendelsohn and Martins, 2018; Pröpper et al., 2015; Revermann et al., 2018; Schneibel, 2013; Schneibel et al., 2017a; Schneibel et al., 2017b, Stellmes et al., 2013; van Wilgen et al., 2022). Their work has been made possible through organisations such as the HALO trust, who are demining active landmines across Angola (Forbes et al., 2015), many of which are located in south-east Angola where conflicts occurred during the Angolan civil war between the Movimento Popular de Libertação de Angola (MPLA) and the União Nacional para a Independência Total de Angola (UNITA) (Brinkman and Alessi, 2009).

Warfare and armed conflicts impact land systems through diverse processes (Baumann and Kuemmerle, 2016). These impacts can be drastic and often long-lasting (Machlis and Hanson, 2008). Fatalities, casualties, human rights abuses, destruction of livelihoods and displacement of those fleeing conflict zones are among the immediate effects on people. Equally, biodiversity loss, fragmentation and alteration of natural ecosystems is common, as observed in Mozambique and the Democratic Republic of Congo (Butsic et al., 2015; Daskin et al., 2018; Nackoney et al., 2014; Stalmans et al., 2019), though nature and wildlife may benefit where human pressures decrease due to armed conflict (Gorsevski et al., 2012; Stevens et al., 2011). Studies in tropical forested conflict zones have shown that forest extent and structure can be affected before, during or after periods of conflict (Basnet and Vodacek, 2015; Butsic et al., 2015; Gorsevski et al., 2012; Nackoney et al., 2014) due to direct and indirect impacts. Direct impacts of conflict on forested ecosystems can include forest loss from bombing, tactical herbicides and intentional lighting of forest fires (Daiyoub et al., 2023; Chandler and Bentley, 1970; Martin, 2023). Indirect impacts of conflict may be associated with long-term alterations of land use or settlement patterns and can include abandonment and avoidance of particular areas due to unexploded ordnance (Henig, 2012; Landsberg et al., 2006), or timber harvesting or conversion of forest to agriculture by displaced people (Hagenlocher et al., 2012; Machlis and Hanson, 2008). During the Angolan civil war, a lack of job opportunities pushed thousands into the informal economy. In provincial capitals, displaced people supplied cheap labour for local landowners to collect firewood and were involved in petty trading at small-scale markets (UN, 2002). Migrations of both internally displaced people to urban areas and returnees moving back to their villages in the last ten years of the war from 1992 to 2002, and the increasingly large numbers of displaced people, particularly in the last four years of the war may have had an impact on the miombo woodlands of southeast Angola.

Global land cover maps perform poorly over the Angolan Highlands (Buchhorn et al., 2020). Previous studies mapping woody vegetation and its change in south-east Angola have either been global, regional for southern Africa, very local in scale or have focussed on mapping complete removal or growth of woody vegetation whilst omitting canopy opening or closure, which are important land cover changes in the miombo ecoregion as both can represent degradation of some ecosystem services (Hansen et al., 2013; Loft et al., 2024; McNicol et al., 2018, 2023; Schneibel et al., 2017a; 2017b). Given this past work, the extent and intensity of recent anthropogenic pressures on the woodlands of the Angolan Highlands is unclear. In more populated areas near Menongue and Chitembo, there was a rise in agricultural expansion and development of an agricultural frontier after the end of the civil war, followed by a later deceleration of cropland expansion (Loft et al., 2024; Schneibel et al., 2017a).

Additionally, sequential waves of degradation followed by deforestation have been observed radiating from major markets into wooded ecosystems over time elsewhere in the miombo ecoregion, and although we know that features such as roads, urban centres, markets and population density affect land cover change dynamics, this has not been quantified for the entirety of the Angolan Highlands (Ahrends et al., 2010; Mertens and Lambin, 2000, 1997; Meyfroidt et al., 2013; Schneibel, 2013).

Our first objective was to examine whether the conclusion of the Angolan civil war affected land cover change, particularly in the woodlands in and around the Lisima region. To accomplish this, we developed a harmonised time series of land cover classifications derived from Landsat imagery from 1990 to 2020. We examined the temporal trends in land cover change to identify changes in woodland extent and structure over time. Our second objective sought to discern spatial patterns and infer the main drivers of land cover change across Lisima and the surrounding landscapes. We aimed to assess if spatial patterns of land cover change were characteristic of land system dynamics such as resource use frontiers, core woodland habitat, or landscapes dominated by regrowth. In addition, we attributed observed changes to a number of drivers based on their relationships with a set of covariates.

Methods

Study region

The Angolan Highlands lie in the south-east of Angola and are the headwaters of the Okavango River, the southern source of the Congo river and the western source of the Zambezi (Lourenco and Woodborne, 2023). Our study area, which covers Lisima (and some of its surroundings to provide spatial context), expands from a less populated highland region within Moxico to the more populated lowlands of the surrounding provinces, totalling 259,024 km^2 (Fig. 1). Several major cities fall within the study area. Huambo, the third most populous city in Angola

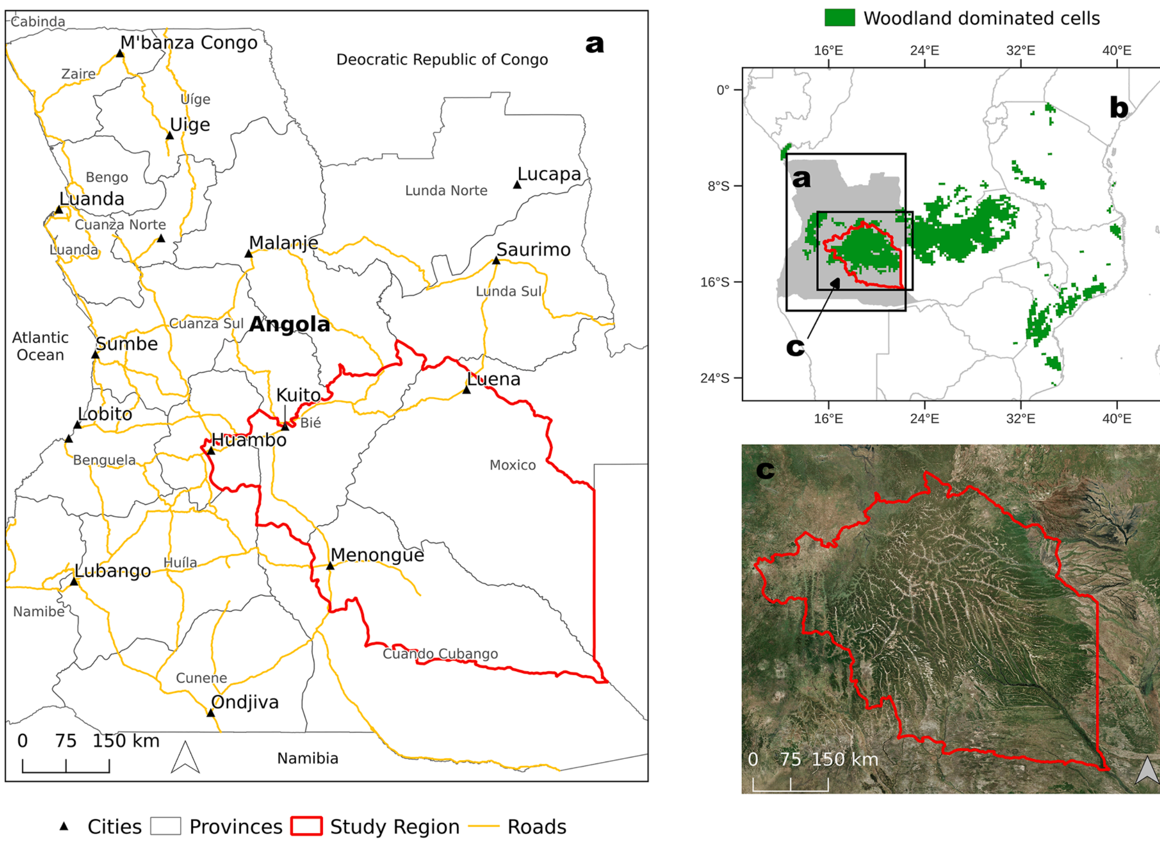


Fig. 1. a) Location of the study region within Angola with roads shown in orange, showing that much of the study region is disconnected from the country's infrastructural network. b) Map centred over the miombo ecoregion where each 400 km2 cell is shaded green if over 75 % of that cell is woodland or forest. Woodland was defined with an NDVI threshold of 0.4–0.6 using Sentinel −2 imagery from 2018 to 2022 c.) High-resolution satellite imagery from Bing showing the study region.

(595,304 people in 2014) and capital of Huambo province, is at the very west of the study area. Menongue (251,000 people in 2014) in the southwest of the study area is the capital of Cuando Cubango province. Kuito (355,000 people in 2014) in the northwest of the study area is the capital of Bié province. Luena (274,000 people in 2014) in the north of the study area is the capital of Moxico province (INE, 2016). Road networks connect these metropolitan areas. Many of the paved roads and bridges in the region were destroyed in the civil war and rebuilt between 2007 and 2010 (Schneibel, 2013). The major road connecting Luena to Kuito is only partially paved, and secondary roads into Moxico province are dirt tracks that can be impassable by trucks and cars during the rainy season.

Across the region, annual rainfall varies from 900 mm – 1320 mm, with less precipitation in the south (Lourenco and Woodborne, 2023). The climate is seasonal, with hot, wet summers from October to May and mild, dry winters from June to September (Huntley et al., 2019). The soils of the study area comprise Kalahari sands (acidic arenosols) that make up an extensive interior plateau within Angola (Huntley et al., 2019). Slow-flowing meandering rivers emanating from source lakes and numerous peat deposits characterise the headwater region (Lourenco et al., 2022a), which has an elevation of ~1119 m in the southeast of Moxico to ~1676 m near the source lake of the Cuito River.

The vegetation formations of southeast Angola comprise tropical and subtropical grasslands and tree and shrub savannas (Goyder et al., 2018; Huntley et al., 2019). Miombo woodland is the dominant vegetation type in the study area, characterised by the genera *Brachystegia* and *Julbernardia* (Gonçalves et al., 2017; Revermann et al., 2018). In southern Africa, this extensive savanna woodland type covers up to 1969,000 km^2 (Ribeiro et al., 2020) and forms a transitional system between the closed rainforests in central Africa and open semi-arid and arid savannas of southern Africa (The SEOSAW Partnership, 2021). The miombo ecoregion supports ~8500 plant species, 4600 of which are endemic (Ribeiro et al., 2020; White, 1983).

Land cover and land cover change classification

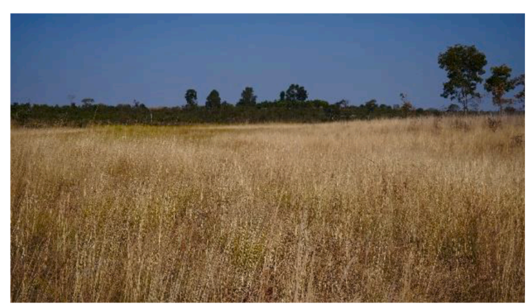
Class identification

We identified several land cover classes that represented the landscapes of southeast Angola (Table 1). This was done based on previous land cover maps, field observations whilst driving through the study area, and high-resolution satellite imagery from Google Earth. These classes were cross-referenced with literature from the region to give the class names and descriptions in Table 1 (Gomes et al., 2021; Goyder et al., 2018; Mendelsohn and Martins, 2018; Revermann et al., 2018; Torello-Raventos et al., 2013).

Remote sensing data. Landsat image stacks of cloud-free dry season imagery (May - October) covering the region were collected in Google Earth Engine for each year in the time series using only images with less than 50 % cloud cover- 7471 images in total between 1990 and 2020, spanning Landsat 4, 5, 7 and 8. 1990 was chosen as the start date to balance the need for a long time series with the increasing sparsity of the early Landsat archive. We then applied cloud masking and terrain correction functions to the imagery. Median dry season reflectance composites were also calculated for each year, for each band.

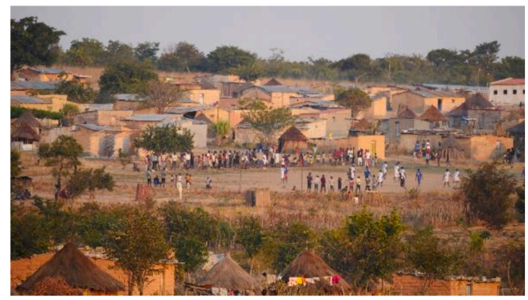
Input feature generation. We tested the utility of different input feature sets by repeatedly classifying a subset of the region with the same training data and comparing classification accuracies. This process revealed the most robust combination of inputs to be the Enhanced

Table 1
Definitions and descriptions of land covers. Woody and grassland land cover terminology follows Torello-Raventos et al., 2013 and Gomes et al., 2021. Photos were taken by C Andrews on field trips to Angola in 2021 and 2022.

Class	Description	Photo
Dense Woodland and Forest	The densest formations of tree cover with closed tree canopies. Mostly miombo woodland, can include shrub-rich forest and other dry forest types.	
Woodland	Open tree canopy formations of miombo woodland, with some grass, including shrub-rich savanna woodland and tall savanna woodland.	
Open Savanna	Where there is the presence of a dominant grass layer and more open tree cover of shorter trees, i.e. long grass savanna. May include shrub savanna.	
Grassland 1	Suffrutex grassland with a sparse layer of geoxyles, usually on sandy soils. Subsequent field validation and cross referencing with Gomes et al. (2021) suggested the geoxyle layer is dominated by *Parinari capensis.*	
Grassland 2	Suffrutex grassland with a sparse layer of geoxyles. Field validation and cross-referencing with Gomes et al. (2021) suggested the geoxyle layer is dominated by *Brachystegia russelliae* on ferralitic soils.	

(*continued on next page*)

Table 1 (*continued*)

Class	Description	Photo
Cropland	Where crop fields are dominant, can include individual fields and clustered field arrangements.	
Settlements	Villages, towns, and cities.	
Water and Riverine Grasslands	Water and seasonally flooded grasslands.	
Bare	Land cover is dominated by bare earth.	

Vegetation Index (EVI), Normalised Burn Ratio (NBR), and Normalised Difference Water Index (NDWI) in addition to greenness, brightness and wetness tasselled caps (see Table SI2 for the full list of input features). EVI outperformed the Normalised Difference Vegetation Index (NDVI) as previously observed in tropical savannas (Abreu et al., 2017; Rosan et al., 2019). NBR and NDWI consistently outperformed other similar spectral indices.

Classification

Training data. An initial land cover classification for 2020 was produced with a supervised Random Forest classifier to allow subsequent proportional-to-area allocation of training labels. Training labels for this initial LCC ($n = 500$) were assigned based on manual interpretation of high-resolution imagery from Google Earth using Collect Earth (Bey et al., 2016).

The final LCC for all years was then produced using a new set of training data and the same method. The new training data ($n = 3000$) were collected proportional to the area of the initial LCC map. Sampled pixels were used only if they remained stable throughout a given time period. We collected training data over three periods- 1990–2000, 2000–2010, and 2010–2020.

This two step process allowed a balanced sample of training data to be used. We used the final training dataset to train a Random Forest classifier, a common and easy to parameterise machine learning approach that is robust to non-parametric data (Breiman, 2001), implemented in Google Earth Engine. We classified each year individually with the appropriate training data from the relevant time period. The classifier was set to multi-probability output, which calculates the probability of each pixel belonging to any given map class, providing an estimate of confidence in land cover classification. We then created additional training data ($n = 500$) in problematic areas with obvious misclassifications (grassland and cropland, and clusters of urban pixels) at a higher intensity to reduce classification errors.

Hidden Markov model. Independently classifying each year of a time series is liable to cause substantial map errors for year-on-year change, and such errors may account for larger areas of change than true land cover change (Abercrombie and Friedl, 2016). We applied a 'Hidden Markov Model' (HMM) to harmonise the land cover time series and reduce map errors.

The HMM assigns an *a priori* probability of land cover transitions between each pair of land cover classes based on a pre-defined matrix (Table SI3). In this case, the matrix assigns low probabilities to all transitions apart from stable land covers, e.g. from woodland to woodland. The HMM combines the a priori transition probability with probabilistic mapped classifications to give a final output of land cover. Transitions are only recorded where sufficient evidence is built up, either where a transition is considered probable (according to the matrix) or where repeatedly allocated as high probability from the classified images produced by the Random Forest classifier. The output of the HMM is a time series that is more consistent through time than independently classified images, with fewer random fluctuations between class values.

Change map extraction. To identify land cover change, we identified where pixels transitioned from one land cover class to another throughout the time series. In our analysis, we focus on change classes relating to the dynamics of wooded pixels. Table 2 explains all change classes included in our analysis, Table SI4 details all possible change classes and shows those which we excluded due to poor user accuracy (< 40 %) or being outside of the scope of this study. Here, we use the terminology "canopy opening" for change classes that represent the opening of woody vegetation - avoiding the negative connotation that "degradation" implies; whilst such opening may lead to lower carbon stocks (McNicol et al., 2018, 2023), it can also lead to a richer and more productive understory. When we use the term "degradation", it is in the context of land use, where management, or a lack thereof, may have led to the diminishment of the ecosystem's ability to provide goods and services (FAO, 2015).

Accuracy assessment. Accuracy assessments based on the sampling strategy outlined by Olofsson et al. (2013, 2014) were carried out with test data collected in the same manner as the training data (above). We performed separate accuracy assessments for land cover maps for 1990, 2000, 2010 and 2020 and land cover change maps in all three-year 'epochs', e.g. 1991–1993, 1993–1995 …, 2018–2021. For each assessment of static land cover, we used 1000 sample points. For change assessments, we used 500 points.

Table 2
Definitions of land cover change classes used in this study.

Change Class	Change type	Description
Dense woodland and Forest to Cropland	Deforestation	Clearing of dense woodland and forest to cropland.
Woodland to Cropland	Deforestation	Clearing of woodland to cropland.
Dense Woodland and Forest to Woodland	Canopy opening	Transition of dense woodland and forest to woodland. Represents a form of tree canopy opening.
Woodland to Open Savanna	Canopy opening	Transition of woodland to open savanna. Represents a form of canopy opening.
Dense Woodland and Forest, Woodland or Open Savanna to Grassland 1 or 2	Canopy opening	Where woody classes (dense or open woodland or savanna) have transitioned to any type of grassland from Table 1.
Cropland to Open Savanna, Woodland or Dense Woodland and Forest	Vegetation regrowth from cropland	Where cropland has regrown to any woody class, e.g. open savanna, woodland or dense woodland and forest. Shows gains in canopy cover.
Grassland 1 or 2 to Open Savanna, Woodland, or Dense Woodland and Forest	Canopy closure	Growth of woody vegetation on formally grassy-dominated pixels. Transition from any grassland class to open savanna, woodland or dense woodland and forest.
Open Savanna to Woodland or Dense Woodland and Forest	Canopy closure	Transition of open savanna class to woodland or dense woodland and forest.
Woodland to Dense Woodland and Forest	Canopy closure	Transition of woodland to dense woodland and forest.

Temporal analysis of land cover and land cover change

Our first aim was to assess whether the conclusion of the Angolan civil war affected land cover change in the woodlands of the study region. To do this, land cover change classes were aggregated into deforestation, canopy opening, canopy closure and vegetation regrowth from cropland (Table 2). Confusion matrices from the accuracy assessments were used to calculate error-adjusted areas for static land covers for each decade and the aggregated change classes for each three-year epoch (Olofsson et al., 2014). We plotted these error-adjusted areas to identify change over time (Figs. 2 and 3) and calculated change rates proportional to the area of the origin land cover class identified in Table 2.

Spatial analysis and syndromes of land cover change

We also aimed to assess if spatial patterns of land cover change were characteristic of land system dynamics (Ahrends et al., 2010). We looked to identify if there were "syndromes" of land cover change which showed a similar mixture of land cover change extents. Syndromes refer to archetypical patterns or processes in the context of land cover and land use change (Lambin et al., 2003; Meyfroidt et al., 2018). These patterns of change may occur at a larger landscape scale and can indicate social, economic, political and ecological dynamics. For example, where high deforestation rates are detected over a large area, an "overexploitation syndrome" might be an appropriate archetype, where socio-ecological processes determine timber extraction (Lambin et al., 2003)

We visually interpreted general patterns of change from the extracted change maps (Figures SI3-SI6) in QGIS 3.10.9 (QGIS, 2023). To identify syndromes of land cover change, we calculated land cover change rates within 10 km by 10 km grid cells for deforestation, canopy opening, canopy closure and vegetation regrowth from cropland. An unsupervised Random Forest classification using these change rates was performed over the study area to produce the 'syndrome classification' for each grid cell. Mean rates for each syndrome were summarised with descriptive statistics post-classification to identify syndrome characteristics.

Correlates of land cover change

To understand the drivers of change and explain the observed syndromes, we used a set of covariates – candidate variables that can be indicative of different drivers of change in the land system. Data for putative correlates of change were collated from publicly available datasets. We used Open Buildings v1 (Sirko et al., 2021) as a proxy for population density and summed for every 3 km^2. Urban centres were also identified from this layer. Major markets were identified using a threshold value of 0.15 for the relative wealth index of settlements (Chi et al., 2022). We used Open Street Map to identify primary, secondary and tertiary roads (Open Street Map, 2023). The woodland edge was identified from our land cover classifications. We calculated Euclidean distance from urban centres, markets, roads and to the woodland edge (separately for both pixels inside and outside the woodland). Fire frequency was calculated from a time series of MODIS burned area data from 2001 to 2019 (Giglio et al., 2006). We then filtered these data layers to exclude densely populated areas with building densities greater than 800 per km^2.

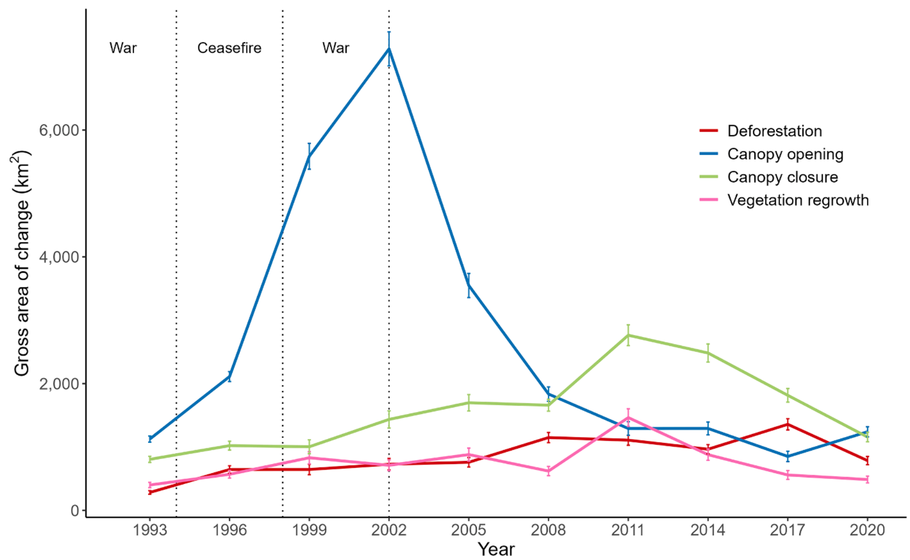


Fig. 2. Error-adjusted areas with standard errors of aggregated change classes for 3-year time steps from 1990 to 2020. The wartime period before 2002 consists of two periods of conflict (up to 1993 and 1998–2002) and a ceasefire period following the Lusaka Protocol (1994–1998). Standard errors were calculated using the methods of Olofsson et al. (2013).

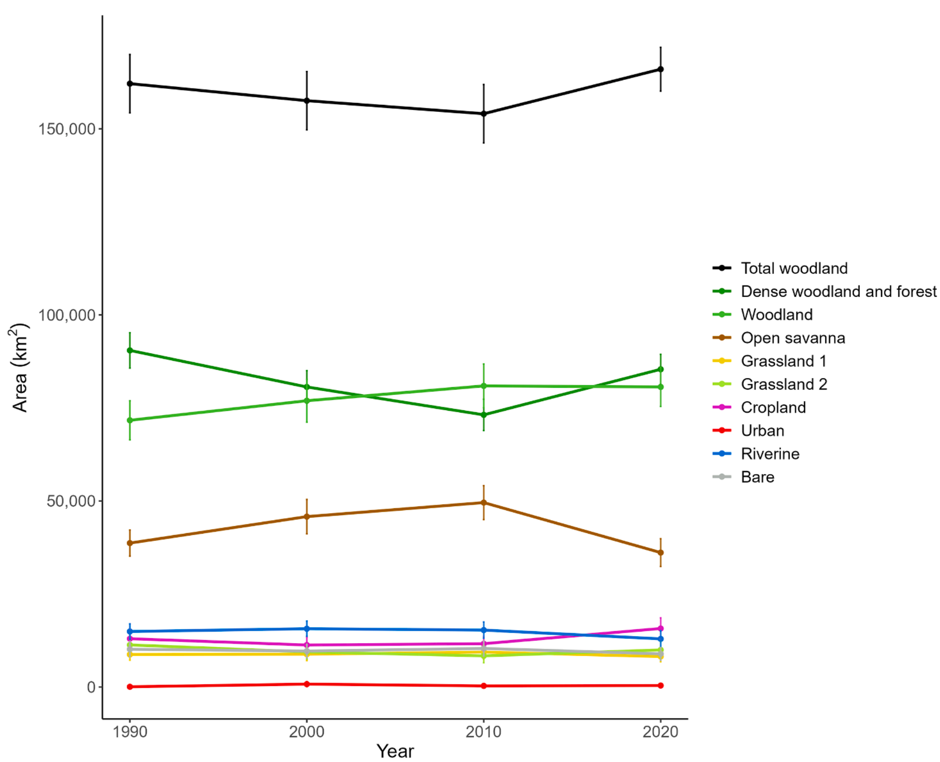


Fig. 3. Error-adjusted areas with standard errors for land cover classes over time. Total woodland was calculated as the sum of dense woodland and forest and woodland classes. Standard errors calculated using the methods of Olofsson et al. (2013).

To assess the effect of the covariates on deforestation, canopy opening, vegetation regrowth from cropland and canopy closure, we fitted Random Forest models for each aggregated change class with the presence of the change as the binary response and the putative correlates as explanatory variables. Partial dependence plots (PDPs) were used to assess the non-linear relationships between land cover change and the explanatory variables.

To assess the overall importance of each covariate, we examined its prevalence as well as the effect sizes revealed by the PDPs. To do this, we used the fitted RF models to predict landcover changes under a scenario of the covariate being twice as high or low as in the observed data set. The predictor was doubled if it was positively associated with the land cover change of interest and halved if it was negatively associated. By comparing this scenario of "doubled drivers" with the predicted response of the observed drivers, we assessed the overall importance of each driver. This allows a more holistic interpretation than e.g. effect sizes (which say nothing about the prevalence of that driver) or relative importance scores of the RF model (which speak to explained variance rather than real-world prevalence).

Results

Temporal patterns of LCC

We identified canopy opening as the dominant type of change throughout the 30-year time series, though large areas of canopy closure were detected following the peak in canopy closure (Fig. 2). The annual area of canopy opening peaked at 3.6 ± 0.1 %/yr towards the end of the Angolan civil war. The area of canopy closure (quantified on an annual basis) increased until 2011. By the 2006–2008 epoch, canopy closure exceeded opening, peaking at 2 ± 0.1 %/yr in the 2009–2011 epoch before declining, suggesting that it was at least partly a response to prior opening. In total, subsequent canopy closure offset 61 ± 2 % of all canopy opening.

Deforestation increased gradually throughout the time series and was often associated with subsequent regrowth. From 1990–2020, the gross area of deforestation nearly doubled (178 ± 36 %), but this did not seem to be influenced by the timing of the civil war. Much of the deforested area was revegetated, as seen by the increase in vegetation regrowth from cropland over most of the time series in Fig. 2. As such, net deforestation was modest: −0.1 ± 0.1 %/yr in the war period and 0.8 ± 0.2 %/yr in the post-war time period of the study. The gross area of deforestation and canopy closure during the civil war period of 1990–2002 was significantly different to that in the post-war period (2003–2020), though canopy opening and secondary vegetation growth from cropland show no such differences- Table 3.

There is little evidence that the end of the Angolan civil war and the subsequent recovery period substantially impacted the extent of total woodland cover in the Lisima region. According to our error-adjusted data, total woodland cover did not change over the time-series (2 ± 3 %, Fig. 3). However, structural changes within the woodland are evident, with an overall 6 ± 3 % decrease in dense woodland/forest and a 12 ± 6 % increase in woodland from 1990 to 2020. The final time step from 2010 to 2020 shows a pronounced increase in dense woodland and forest and decrease in open savanna, signifying possible changes in trajectories compared to preceding time steps.

Spatial patterns and syndromes of LCC

The change maps (Fig. 4) show land cover change radiating from numerous features across the study region, such as major markets, urban centres and roads. This expansion is evident over time, with canopy opening advancing ahead of deforestation, and vegetation regrowth taking place closest to markets and some urban centres. In addition to this pattern manifesting at local scales (Fig. 4c), we observe this pattern emerging across the whole study area (Fig. 4b).

The unsupervised Random Forest classification of change rates identified five major syndromes, which we names according to their dominant LCC rates (Table 4). These syndromes identified a clear woody resource-use frontier consisting of a “deforestation frontier” syndrome, a “mixed woody losses” syndrome, and an “overharvesting frontier”, which permeates into an identified “core” syndrome, an area of woodland that exhibits relatively moderate rates of canopy opening and closure. We also identified a “woody encroachment” syndrome which identifies areas of prevailing canopy closure (Fig. 4). Other syndromes occupying less than 2 % of the area or that were highly populated with little woodland cover were aggregated to the “Other” class.

Table 3
Mean gross areas of change for each change class during the wartime period of the time series (1990–2002) and the post-war period of the time series (2003–2020) ± standard error of the mean.

Change class	War period (km^2)	Post-war period (km^2)
Canopy opening	2940.5 ± 1352.9	2476.8 ± 867.6
Deforestation	524.1 ± 120.6	978.5 ± 89.8
Canopy Closure	943.1 ± 69.3	1858.5 ± 215.3
Secondary vegetation growth	599.7 ± 125.0	799.7 ± 124.4

Both canopy opening and closure are pervasive across the region (Fig. 4 and Table 4). Core woodland was identified as the syndrome with the largest areal extent (52 % of the classified area) and has a 16 ± 3.2 % canopy closure rate and a 19 ± 2.4 % canopy opening rate. This syndrome identifies the “core” area of woodland in the region. The second most common syndrome (15 % of the classified area) is the overharvesting frontier, which permeates into the core extent of woodland (Fig. 4a), where the mean canopy opening rate is double that of the mean canopy closure rate (Table 4). This syndrome identifies prevailing net losses of woody vegetation and is likely to represent a reduction in woody biomass.

Syndromes with high deforestation rates have smaller extents. The deforestation frontier, dominated by a relatively high deforestation rate, occupies only 2 % of the map and is concentrated towards the more populated parts of the study area. Between the deforestation frontier and the overharvesting frontier, the “mixed woody losses” syndrome forms an intermediate zone of land cover change that has predominantly experienced woody losses from both deforestation and degradation.

Correlates of LCC

Deforestation was largely associated with urban areas, roads, fire (Fig. 5) and distance to the woodland edge. The predictions from the percentage change model (Fig. 6) showed that halving the distance to urban centres had the most impact on the deforestation rate, increasing it by 18 %. The PDPs (Fig. 5) show that urban centres have the strongest association with deforestation within 0–20 km. They also show that the effect of roads on deforestation is greatest within 5 km of the road and within 40 km of markets.

Canopy opening was most associated with fire count, distance to the woodland edge and distance to markets. Doubling the fire count causes the largest change in the predicted rate of canopy opening (+13 %), followed by halving the internal distance to the woodland edge (+10 %), halving the distance to markets (+7 %), and halving the distance to roads (+5 %). The PDPs show that the odds of canopy opening are highest at around ten fire events, however fire frequencies this high are rare within woodlands and forests. The largest effect of internal distance from the woodland edge is within the first 0.5 km (figure SI8). Distance to roads shows a clear negative relationship with the odds of canopy opening, and there is a large negative effect of distance to road on canopy opening within the first 20 km away from urban centres.

Canopy closure appears to be closely linked to fire, with the PDPs and the percentage change predictions showing a large negative impact of fire frequency on canopy closure rates. Distance to woodland edge was also important for canopy closure which might also be related to fire regime. The covariate with the largest predicted effect on canopy closure according to the percentage change model was fire count. Halving the incidence of fire increased the rate of canopy closure by 26 %. This was followed by halving the external distance from the woodland edge (+ 14 %) (Fig. 6). The PDPs show that the odds of canopy closure are highest between 0 and 3 fires and in remote areas away from roads.

Vegetation regrowth after being cropland was mostly associated with fire, distance to markets and distance to roads. Halving the fire count increases the rate of regrowth on former cropland by 8 %. Doubling the distance to markets and roads increases this rate by 3 %. The PDPs show that the odds of vegetation growth are highest between 60 and 100 km from markets and further away from urban centres. We also observed that vegetation growth from cropland is more likely below ten buildings per km^2 (figure SI10).

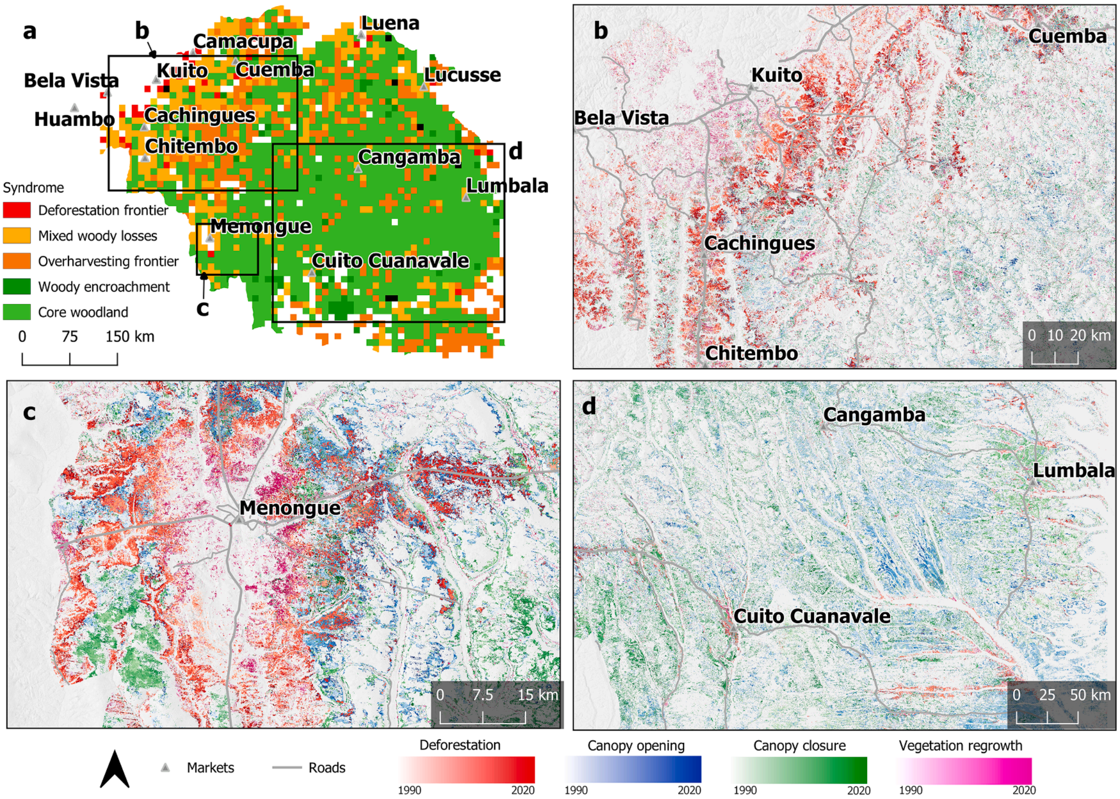


Fig. 4. Syndrome map and land cover change maps for three areas of interest. The colour scales identify time of detected change, the darker the colour the more recent the change. a.) Syndrome classification showing frontier areas and core woodland. b.) Frontier area with widespread deforestation and canopy opening advancing from Kuito and other urban centres. c.) Deforestation and canopy opening radiating out from Menongue. d.) Widespread canopy opening and closure in the core woodland area. See supplementary information for individual change maps. Topographic basemaps from Esri (2012).

Table 4
Summary of syndromes produced from the unsupervised Random Forest classification of the land cover change (LCC) data. Mean rates of change of each land cover change are given as a percentage of the study area and are for the full time series ± shows standard deviation of the LCC rate within each syndrome.

Syndrome	Deforestation rate (%)	Canopy opening rate (%)	Canopy closure rate (%)	Vegetation regrowth rate (%)	Percentage of study area (%)
Deforestation frontier	29 ± 0.8	18 ± 13	11 ± 2.9	6 ± 1.5	2
Mixed woody losses	16 ± 0.1	22 ± 3	14 ± 3.3	4 ± 2.1	14
Overharvesting frontier	2 ± 1.4	21 ± 2.2	10 ± 0.9	1 ± 0.4	15
Woody encroachment	0 ± 0.3	5 ± 3.2	22 ± 8.6	0 ± 0.1	2
Core woodland	2 ± 2.8	19 ± 2.4	16 ± 3.2	1 ± 0.6	52
Other	11 ± 4.8	9 ± 4.8	12 ± 6.1	2 ± 1.5	14

Discussion

Overall, our results show that the conclusion of the Angolan civil war resulted in an immediate increase in canopy opening, likely related to the movement and resettlement of displaced people. These losses in woody vegetation were largely offset by subsequent canopy closure in the following decades. Whilst structural change in the woodland is evident as a legacy of the war, we did not find that total woodland cover had significantly changed.

We observed woody resource use frontiers in our land cover change maps and syndrome classification that identify not only a deforestation frontier, but also an overharvesting frontier and an intermediary zone of deforestation and intense harvesting. From our PDPs and covariate importance assessment we identify urban demand as the likely driver of these changes. Canopy closure however, shows no clear relationship with anthropic covariates besides fire, therefore we attribute this growth class to biophysical drivers. Finally we identify areas of “core woodland”, where no significant net change in woodland was detected, despite considerable gross changes.

Temporal patterns of change in relation to the Angolan civil war

In the 1990s, the final decade of the Angolan civil war, increasing numbers of people moved away from conflict zones, causing changes in landcover dynamics across the region. Throughout Angola, 4.1 million internally displaced people sought refuge in camps and urban areas during wartime, the vast majority with host families in urban settings (Lari, 2004). This shift in population distribution from a majority rural population to 60 % of Angolans living in urban settings occurred over just three decades (UN, 2002), increasing pressure on woody resources in proximity to urban centres.

Canopy opening was the largest change class throughout the 30-year time series, peaking around the end of the Angolan civil war in 2002. However, this is not a net change; 61 % ± 2 % of canopy opening was offset by later canopy closure, which became the largest change class after 2010. This is likely to have led to negative transient outcomes for carbon storage and some aspects of avian and mammalian biodiversity (Chidumayo, 2013; Tripathi et al., 2019), yet was likely a positive outcome for the livelihoods and well-being of woodland resource-dependant individuals in the post-war period. The observed

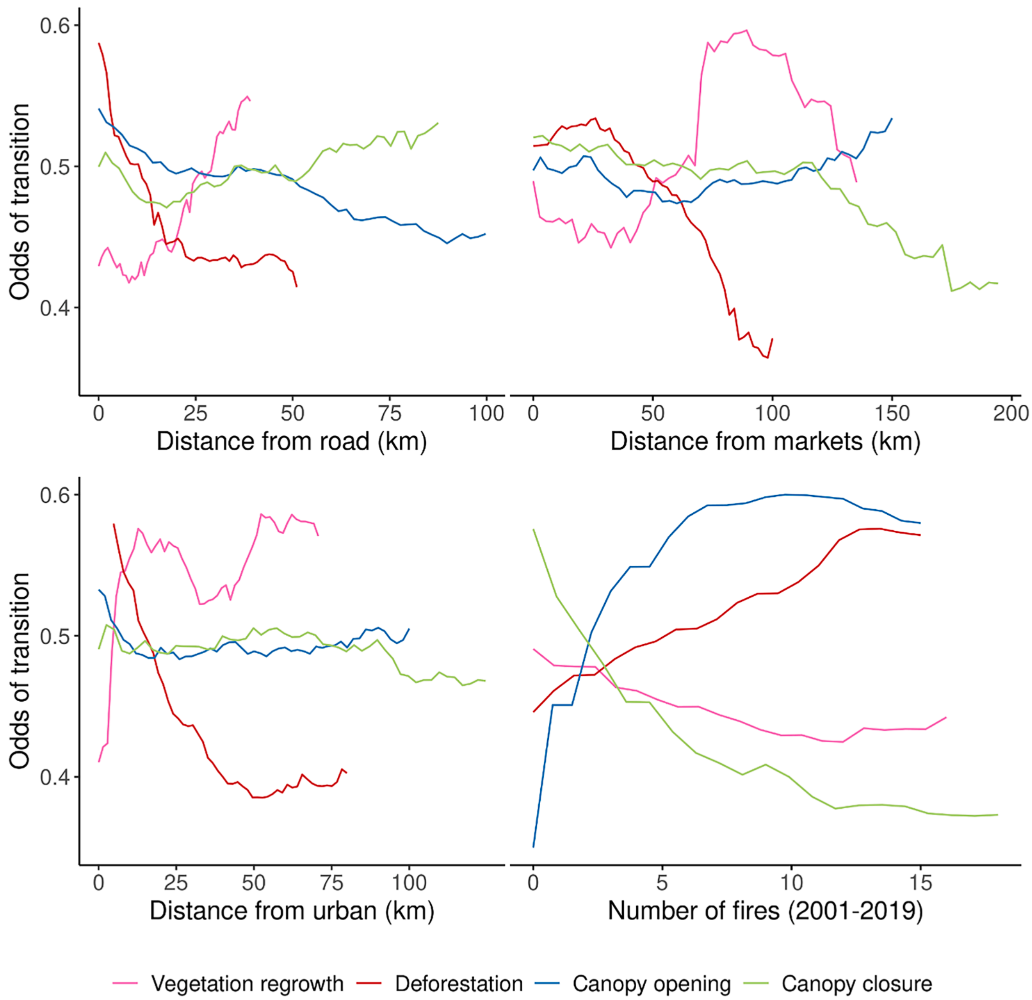


Fig. 5. Partial dependence plots from the Random Forest models for each change class showing the odds of land cover transitions against the covariates that were most associated with land cover change, see figures SI7-SI10 for the full set of partial dependence plots.

peak in canopy opening (Fig. 2) could be attributed to the use of woodland resources by displaced people, in addition to people returning towards the end of the conflict (Brinkman and Alessi, 2009; Catarino et al., 2020; Gaynor et al., 2016). Whilst we do not have robust longitudinal population data, known patterns of rural to urban migration in the war and post war period fit these conclusions (Lari, 2004; UN, 2002).

Canopy closure after 2008 and maintenance of total woodland cover indicates a recovery of woody biomass and longer-term positive outcomes for carbon storage and biodiversity in the post-war era of the study period. Canopy closure in the post-war period, almost a decade after the peak in canopy opening, signifies the legacy effects of the civil war (Fig. 2 and Table 3), which can persist well after a conflict ends (Nackoney et al., 2014). Moxico and the surrounding provinces were the epicentre for some of the heaviest fighting in the civil war and still harbour active landmines. Although the locations of these landmines are largely known and are actively being cleared, they still pose a risk to potential land users. Land contaminated with landmines disincentives land use and presents an opportunity for woodland regeneration (Baumann and Kuemmerle, 2016). Cuando Cubango province, in the south of the study region, was once dubbed "the lands of progress" by state authorities, and parts of this province, like more populated regions of Huambo and Bié, have experienced growing infrastructural development in the decades since the war. However, much of the south-east of Angola remains marginalised by the central government (Brinkman and Alessi, 2009) due to past affiliations with UNITA. The marginalisation of these remote rural communities is associated with little infrastructural development, limiting driving forces of land cover over large areas and maintaining much of the region's 'remoteness'. These legacy effects of the Angolan civil war have likely contributed to the peak in observed canopy closure and have facilitated processes of woodland recovery.

We observed the structure of the woodland to be quite dynamic over the 30 years, with fluctuations between dense woodlands and forest and woodlands in some areas, even in the 'core woodland' area with less pronounced human influence. This core woodland, far from being a static unused landscape, has moderate rates of canopy opening and closure, which might be associated with shifting cultivation, fire and other land uses. This pattern of utilisation and (re)growth has been highlighted across the miombo ecoregion and is common even in the most remote areas (Chidumayo, 2013; McNicol et al., 2018). Miombo ecosystems have evolved to be resilient to disturbance from fire and herbivory (Kissanga et al., 2024; McNicol et al., 2015; Ribeiro et al., 2020; Ryan and Williams, 2011) and evidence from other areas of the miombo ecoregion shows that tree species diversity in regrowing plots can recover to that of mature woodlands by 10 years after agricultural abandonment and above ground carbon stocks are predicted to reach equivalent values to mature woodlands within 30–35 years (Chidumayo, 1991; Godlee et al., 2020; Gonçalves et al., 2017; Kalaba et al., 2013; McNicol et al., 2015). It is likely therefore that most of the observed canopy closure in this study is an outcome of the key ecological characteristic of resilience in these systems.

We found that deforestation did steadily increase for the whole region over the 30 years, but we did not find a dramatic increase

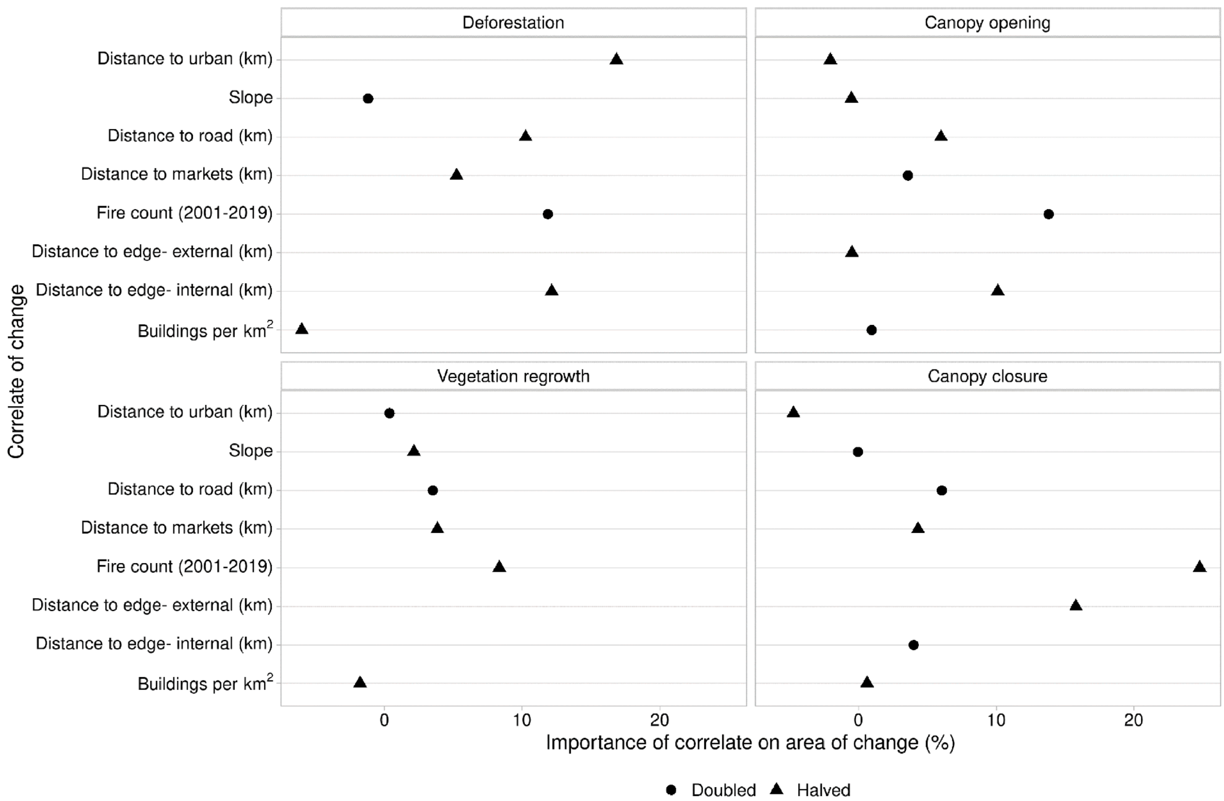


Fig. 6. Importance of each covariate on land cover change. Importance is expressed as the percentage change in predicting land cover transition extent after doubling or halving a covariate. Predictors were doubled if positively associated with land cover change or halved if negatively associated.

immediately after the war, as can be seen around more populated areas (Loft et al., 2024; Miapia et al., 2021; Schneibel et al., 2017a). Longer-term resettlement of displaced peoples, population growth and increased urban demand could have driven this steady expansion in cropland over the study period, resulting in a significant difference in the gross area of change in the post-war period compared to the war time period (Table 3). However, the remoteness and inaccessibility of the woodlands of Moxico and poor soil conditions limit their feasibility for commercial agriculture (Mendelsohn and Martins, 2018), hence why we may not have detected a similar rise in cropland across the whole region.

Spatial patterns, causes and drivers of change

Deforestation and canopy opening

Our results identified clear patterns of land use as defined by von Thünen's et al. (1966) theory of spatial organisation of land use, in addition to sequential waves of canopy opening advancing ahead of deforestation, emanating from major markets, urban centres and some roads (Ahrends et al., 2010). Deforestation and canopy opening were likely driven by urban demand. The covariate importance analysis identified urban centres as having the largest impact on deforestation. Halving the distance to urban centres increased the deforestation rate by 18 %. Similarly, the PDPs showed that deforestation is more likely closer to urban centres, markets and roads. The importance model and the PDPs showed that fire count was the most important correlate of change for canopy opening. Like deforestation, the partial dependence plots show that canopy opening is more likely closer to markets (within 60 km), urban centres and roads (both within 20 km).

A large portion of the northwest of the study area is occupied by the mixed woody losses and overharvesting syndromes, whilst the deforestation-frontier syndrome is rare. Buchadas et al. (2022) show that most of the frontier area in SE Angola is comprised of an active frontier with high woodland cover remaining and emerging frontiers in and around the core area of woodland. We found that deforestation frontier occupied 2 % of the classified area, suggesting high rates of conversion of woodland to cropland at large spatial scales will be unlikely in the near future. Future predictions of agricultural expansion across the region are unclear, though less woodland is expected to be lost in Angola compared to Zimbabwe and Mozambique in future scenarios of predicted yield increases and resulting rebound effects (Pratzer et al., 2023). Whilst deforestation is still occurring at moderate rates in the mixed woody losses syndrome, occupying 14 % of the classified area, higher rates of canopy opening have been detected here, as well as in the overharvesting frontier. In the study region's northern and western areas, these syndromes are associated with agricultural production and resource extraction for urban demand.

The strong association between urban centres and deforestation to cropland that we have observed shows that agricultural expansion is the proximate cause of deforestation, likely driven by urban demand for agricultural commodities. Furthermore, the increased likelihood of canopy opening occurring within 60 km of markets and 20 km of urban centres points to an active charcoal industry in these areas as a proximate cause of this type of land cover change (Ahrends et al., 2010). Again, this charcoal industry is likely driven by urban demand for wood fuel products (Chidumayo, 1989, 1987; Zulu and Richardson, 2013). Anecdotal evidence highlights selective logging, targeting rosewood in the region as another proximate cause of canopy opening, driven by international demand for timber products. These patterns of land cover change focussed around urban centres, markets and roads signify different land uses, where extraction of woody resources at the advancing edge of the frontier removes the most valuable stems and sequential waves of tree harvesting target the next most valuable woody resources, leading to a situation of "logging down the profit margin" (Ahrends et al., 2010; Pauly et al., 1998) until woody resources are cleared, in some cases for agriculture, most typically on the land with the highest rent closest to markets (Angelsen, 2007; von Thünen et al., 1966). Infrastructure to facilitate supply chains between markets and urban centres can increase the likelihood of land cover change events. Our PDPs show that land cover change processes are more likely closer to road networks, which corroborates with the findings of Scheibel (2013) in the southwest of the region.

Vegetation regrowth and canopy closure

Identifying the dynamics of vegetation regrowth following cropland allows us to further understand agricultural practices. We observed growth of vegetation after cropland to be more likely further away from markets and urban centres and at lower population densities. Therefore, in line with the bid rent theory, it is likely that agricultural practices closer to markets are comprised of more permanent forms of agriculture associated with commodity cropping to supply urban demand (Meyfroidt et al., 2014). Further away from markets at lower population densities shifting cultivation is practiced. Shifting cultivation results in the growth of woody vegetation when cropland no longer used. Although often maligned as a degrading land use practice (Mertz, 2009), shifting cultivation creates patchiness in vegetation cover, providing important habitat for select species and increasing β-diversity in the landscape, potentially allowing rare or subdominant species to proliferate, and as previously mentioned, the resilience of miombo ecosystems enables their recovery after disturbance from shifting cultivation (McNicol et al., 2015). The growth and prevalence of disturbed woodlands will also affect local livelihoods and well-being, presenting a potentially important but different suite of provisioning services than the mature woodland it replaces (Padoch and Pinedo-Vasquez, 2010; Pritchard et al., 2019).

Canopy closure showed no clear relationship with urban centres or roads and is almost as likely at all distances from these features. Though odds do increase further from markets, we attribute this to higher fire frequencies in the south-east of the study area far from markets. Fire was the most important correlate of change for canopy closure in the percentage change model, where halving the incidence of fire increased canopy closure rates by 26 %.

Due to the prevalence of canopy closure at all distances from "human" correlates of roads and urban centres and being more likely in remote woodland and at lower fire frequencies, we attribute this transition type to global biophysical drivers such as increasing CO_2 and changing climate (Buitenwerf et al., 2012; García Criado et al., 2020). The decline in elephant populations as a result of mass killings during the civil war (Chase and Griffin, 2011) may also contribute to decreased levels of disturbance in the woodlands that facilitate the densification of woody vegetation in remote areas. In some cases, this increase in woody vegetation takes the form of woody encroachment, where trees expand into formally unwooded systems, often with delatarious impacts on diversity (Abreu et al., 2017; Mogashoa et al., 2021; Smit and Prins, 2015). The identified "woody encroachment" syndrome shows where canopy closure has exceeded all other change rates. In contrast to Loft et al. (2024), we do not find evidence for extensive net woody encroachment as this syndrome only occupies 2 % of the study area.

Fire and LCC

Reduced burning frequency prompts canopy closure, and this is more likely in remote, denser patches of woodland. Biomass of miombo woodlands is maintained where the fire return interval is greater than 5 years and at low burning intensities (Ryan and Williams, 2011). The median fire return interval for the miombo woodlands in the study region between 2001 and 2020 was 4.5 years. However, fires that did occur were predominantly late dry season fires (van Wilgen et al., 2022), which can produce more intense fire events due to increased fuel load in the woodland and potentially cause biomass loss (Saito et al., 2014).

Our results show that canopy opening is most likely in the same pixels as higher fire frequencies and closer to the woodland edge. In this study, we have not analysed the temporal sequence of burning and land cover change, therefore we cannot definitively infer causality. Whilst it may be likely that fire is the causal agent of change in some of these cases, it is also possible that fires are burning after canopy opening occurs due to grass cover and flammability increases, as denser vegetation structures are more difficult to burn (Newberry et al., 2020). Fire frequency, particularly in the woodlands, is highest in the driest part of the study area in the south-east. Fire is an important tool for rural livelihoods in the miombo ecoregion, used for clearing woodland for agriculture, burning grassland to encourage new grass shoots to grow, in hunting, apiculture and charcoal production (Ryan et al., 2016, 2014; Sedano et al., 2020). The largest burnt areas in Angola between 2001 and 2019 were recorded between 2003 and 2005, linked to the resettlement of refugees after the civil war (Catarino et al., 2020; Lourenco et al., 2022b), occurring a year after the observed peak in canopy opening that we observed from our results- presenting a case for fires burning in woodlands once they have been opened by extractive land use activities.

However in the southeast, syndromes dominated by canopy opening are positioned far away from most markets and urban centres yet have strong associations with higher burning frequencies. These dynamics are what we might expect from savanna systems over time in terms of variation in tree cover (Lehmann et al., 2008), but also show that livelihoods within this syndrome have not been a net cause of land cover change despite land use pressures from surrounding woody resource use frontiers.

Conservation implications

Understanding spatial patterns in land cover change is vital for targeted conservation efforts. Addressing the urban demand for agricultural and forest products is crucial for socio-ecological sustainability. Our research highlights the importance of conserving core woodlands, which are dynamic and shaped by local people, *and* frontier regions where land use impacts are more obvious. Frontier conservation avoids immediate biodiversity and carbon losses and can produce areas of high biodiversity value relative to degraded conditions if recovery is possible (Sacre et al., 2019). Buchadas et al. (2023) emphasize the need for conservation policies that meet local community needs and expand protection near deforestation frontiers.

An intuitive approach to identify potential protected areas for biodiversity and carbon benefits in southeast Angola may be to prioritise intact and undisturbed patches of woodland (a 'wilderness conservation' approach). This is commonly combined with 'buffers' from drivers of woody losses, such as urban centres, markets, roads, and areas experiencing frequent burning. Protected areas in these kinds of arrangements have been shown to lower deforestation and degradation rates whilst increasing regrowth rates in African woodlands (McNicol et al., 2023), and evidence from around the globe agrees that protected areas, generally, result in reduced conversion of woodland or forest and higher above ground carbon stocks (Duncanson et al., 2023; Joppa and Pfaff, 2010).

This 'wilderness conservation' approach has the potential to deliver long-term benefits at the cost of frontier woodland loss (Watson et al., 2018) and may induce leakages and rebound effects of cropland expansion in and around the area of conservation interest (Meyfroidt et al., 2022). Some evidence shows that remote protected areas with lower initial human pressure may experience more human pressure after establishment, and deforestation pressure can increase within protected areas once the surrounding forest is lost (Buřivalová et al., 2021; Geldmann et al., 2019). Additionally, formal protection may have the unintended consequence of eroding collective, long-term resource management and lead to loss of economic opportunities, resulting in communities illegally overexploiting resources that were once used sustainably (Adams et al., 2004; Ostrom, 1990).

This dichotomy between 'wilderness' and 'frontier' conservation should move towards a context-specific understanding of the benefits and dis-benefits of each approach and prioritise efforts accordingly

(Sacre et al., 2019). It may be worth questioning whether a protected area is the most suitable strategy to curb woodland conversion in southeast Angola- particularly considering the challenges facing the current protected area network within the country. The Angolan civil war undoubtedly disrupted any progression of the conservation agenda within Angola- the protected area network has shown few signs of recovery since the peace of 2002 (Huntley, 2017). Current protected areas are systematically underfunded, resulting in a lack of equipment and insufficient human resources in terms of both the number and pertinent experience, at the level of individual national parks and the Instituto Nacional da Biodiversidade e Conservação (National Institute for Biodiversity and Conservation). Ultimately, this results in weak law enforcement to address issues such as poaching, illegal extraction of resources, and human expansion into protected areas (Russo et al., 2022).

In terms of our findings, canopy opening, of which a third has been semi-permanent, and could be described as a form of degradation, was the most prevalent change class within the overharvesting frontier syndrome, permeating into the core extent of woodland. Proximate causes of degradation such as timber harvesting and charcoal production are difficult to reduce within protected areas due to the transient nature of these activities, compared to deforestation caused by sedentary activities such as agriculture (Ahrends et al., 2021; McNicol et al., 2023). Other conservation measures, such as land use planning, value chain interventions and sustainable management of natural resource use may be more appropriate to mitigate the impacts of these livelihoods.

We also note the resilience of miombo systems and their ability to recover after disturbance, the remoteness of parts of this region, and the general unsuitability of soils for large-scale commercial agriculture as factors that allow for the maintenance of woodland cover in the absence of a protected area. Nevertheless, external threats to the intactness and functioning of this region may yet increase as growing urban populations in southern Africa's secondary cities (Zimmer et al., 2020), increasing international demand for timber resources, and infrastructure projects (Roque, 2013) fuel the expansion of surrounding woody resource frontiers.

Limitations

Several limitations in our study highlight the need to interpret our results cautiously. First, delineating and classifying vegetation types in tropical dry forests and savannas remains challenging (Torello-Raventos et al., 2013). Producing a land cover classification with hard categorical classes based on medium-resolution space-borne optical imagery may oversimplify tropical dry vegetation types when savanna systems often display subtle changes along structural and floristic continua (Feilhauer et al., 2021). Second, more refined methods for cropland mapping at high spatial resolutions could be used (e.g. Rufin et al., 2022), which would give more robust cropland and deforestation metrics. However, we prioritised maintaining consistent longitudinal data throughout the multi decadal Landsat archive. Third, the land cover change processes derived from our time series of land cover maps were analysed against mono-temporal datasets for correlates of change. Further work could compare land cover change data against longitudinal datasets of covariates of change. Finally, we identified that canopy opening and closure were most likely at higher and lower fire counts respectively, but we did not prove causality of burning events and land cover change here.

Conclusion

We present a land cover change dataset that has allowed a comprehensive overview of land cover change dynamics, their spatial patterns and drivers for the highlands of south-east Angola and surrounding areas for the first time. We observe that the woodlands have experienced structural changes over the 30-year study period, likely due to urban demand for agricultural commodities, wood fuel and timber-derived products. This demand was exacerbated by the Angolan civil war, though canopy closure offset canopy opening by almost two-thirds, demonstrating the resilience of these systems. This Angolan extent of woodland is critically important within the miombo ecoregion for biodiversity, carbon storage, hydrological regulation and the livelihoods of local communities and people and wildlife across the Okavango, Congo and Zambezi Basins. Our findings identify where pressures on the woodlands are most prevalent, which will aid in prioritising conservation interventions and policy decisions. We identify the need to produce agricultural commodities, wood fuel, and timber-derived products sustainably to supply urban populations and avoid woodland degradation and deforestation in the detected frontier regions. If conservationists aim to establish a protected area in line with achieving the 30 % area-based conservation by 2030 target of the CBD, interventions should be based on local context and landscape dynamics. We suggest that further work continues to engage with and empower local communities to secure their rights and land tenure.

CRediT authorship contribution statement

Christopher A. Andrews: Writing – review & editing, Writing – original draft, Visualization, Validation, Project administration, Methodology, Formal analysis, Data curation, Conceptualization. **Samuel Bowers:** Writing – review & editing, Methodology. **Luisa F. Escobar-Alvarado:** Writing – review & editing. **Kai Collins:** Writing – review & editing, Conceptualization. **Kyle G. Dexter:** Writing – review & editing, Supervision. **Casey M. Ryan:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.tfp.2024.100623.

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