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Woody biomass increases across three contrasting land uses in Hurungwe, mid-Zambezi valley, Zimbabwe

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Globally, Miombo woodlands store important quantities of carbon, with tree cover and carbon stocks strongly determined by human use. We assessed woodland cover and aboveground carbon (AGC) stocks of miombo along a utilisation gradient on three different land use types, that is, a national park, a buffer zone and a communal area. Woodland cover and carbon stock changes were assessed through mapping of AGC between 2007 and 2017 using Phased Array L-Band Synthetic Aperture Radar observations (ALOS-PALSAR 1 and 2). Woodland cover was higher in the national park and the buffer zone than in the communal area for both 2007 and 2017. In 2007, mean AGC stock was not significantly different ($p = 0.05$) across all three land use types. However, in 2017, mean AGC was significantly lower ($p < 0.001$) in the buffer zone and communal area than in the national park. In all three land use types, Miombo woodland cover and mean AGC gains outweighed losses over the 10-year period. AGC gains were significantly higher ($p < 0.001$) in the national park than in both the buffer zone and the communal area. Results of the study indicate that woodland cover and aboveground carbon increased in all three land use types despite the observed human disturbance over the study period. Both variables recorded a lower increase with elevated utilisation. The study concluded that sustainable resource utilisation is possible without loss of such ecosystem services as carbon sequestration and climate change mitigation.

Keywords: climate change, disturbance, land use change, resource utilisation, sustainable use, woodlands

Introduction

There is increasing recognition of the role of forests and woodlands in climate change mitigation, including in the Paris agreement of 2015 and, more recently, the *Glasgow Leaders' Declaration on Forests and Land Use* (Pelletier et al. 2017; Nasi 2021). Previous studies have demonstrated that miombo ecosystems have tremendous potential to store carbon and act as a carbon sink (Munishi et al. 2010; Kuyah et al. 2016): Southern African woodlands, including miombo, store between 18.0 ± 1.8 petagrammes of carbon (PgC) and 24.4 ± 2.4 PgC evenly distributed between woody vegetation and the soils (Ryan et al. 2016), comparable to the Congo basin forests. Thus, these woodlands contribute significantly to the global carbon cycle and regulation of climate change.

Miombo biomass is usually linked to edaphic features, precipitation and woodland cover change (Ribeiro et al 2021a). Woody biomass in undisturbed mature Miombo ranges from 30–70 Mg ha^{-1} in the dry miombo stands of Mozambique, Tanzania and Zimbabwe (Ribeiro et al. 2013; Kachamba et al. 2016; Lupala et al. 2017) to 100–144 Mg ha^{-1} in old-growth wet miombo (Kalaba et al. 2013; Gonçalves et al. 2017). The most common disturbance agents in Miombo woodlands are human activities, elephants and fire, all of which interact (Frost 1996; Mapaure and Moe 2009). Clearance of land for cultivation and selective harvesting of trees for various purposes, including fuelwood, charcoal production and construction, are the main human activities that directly affect woodlands (Dewees et al. 2010; Bruschi et al. 2014; Syampungani et al. 2014), often linked to urban and international markets (McNicol et al. 2018). These disturbances eventually result in woodland cover loss (deforestation and forest degradation), with significant losses in aboveground carbon stocks (AGC). For example, Ribeiro et al. (2021a) found that changing the regime from 3.3 to annual fire return intervals resulted in a Miombo woodland shifting from a C sink to a C source in Niassa reserve, Mozambique. In another study, Williams et al. (2008) found that clearance for agriculture reduced stem wood C stocks by 19.0 t C ha^{-1}.

Thus, disturbances in miombo play an important role in determining local biomass variations (Ribeiro et al. 2008a), even though little is known about the impact of human disturbance on the Miombo woodland cover, carbon stocks and their change over time.

In Zimbabwe, Miombo woodlands are found in almost every land use category: national parks, communal and resettlement areas, and commercial farmlands (Nyoka et al. 2011). In national park areas, the Parks and Wildlife Act of 1975 restricts access to forest and woodland resources, hence, due to limited or the absence of human disturbance, national parks are viewed as important areas for conserving carbon stocks and maintaining intact woodland cover (Banda et al. 2006). Conversely, communal areas have few restrictions on utilisation, and communities are allowed to access woodland resources according to the Communal Lands Forest Produce Act of 1988. As a result, where communal areas are adjacent to national parks and their buffer areas, the potential for human disturbance increases with distance from the protected area to communal settlements, thereby creating a disturbance or utilisation gradient (Muposhi et al. 2016; Gotore et al. 2020). Therefore woodland cover and AGC are likely to change along the gradient (Banda et al. 2006; Chinuwo et al. 2010; Muposhi et al. 2016) due to differences in intensity of tree harvesting, tree density and frequency of fires over time (Gotore et al. 2020). Despite this recognition, studies are limited, and the relative impacts of these land use types on AGC and woodland cover are not well measured.

Land use and land cover change information is important in carbon stock and emissions assessment. Studies have shown that agricultural expansion has been an important driver of woodland loss in resettled areas since Zimbabwe's fast-track land reform programme of 2000 (Matavire et al. 2015; Nyelele et al. 2018) which redistributed more than 3 000 commercial farms from white commercial farmers to retrenched farm workers and landless, poor households in overcrowded communal areas characterised by nutrient-poor soils (Scoones et al. 2010). However, abandoned agricultural areas in communal lands are regaining woodland cover (Scharsich et al. 2017). Studies in the region (Williams et al. 2008; Mwampamba and Schwartz 2011; Kalaba et al. 2013; McNicol et al. 2015; Gonçalves et al. 2017) have shown that miombo is resilient, regaining most of its floristic composition and carbon stocks within10–20 years (Ribeiro et al 2021b) after agriculture ceases. For example, aboveground carbon has been estimated to accumulate at about 0.7–0.8 Mg C ha^{-1} year^{-1} in dry miombo fallows (Chidumayo 1990; Williams et al. 2008; McNicol et al. 2018), and 1 Mg C ha^{-1} year^{-1} in wet miombo (Kalaba et al. 2013). While most studies in Zimbabwe have focused on impacts of the fast-track land reform programme and changes in different land use categories, there remains a need to understand woodland cover dynamics along human disturbance gradients given that land use change processes are predicted to drive ecosystems and service provision changes (Ryan et al. 2016).

Generally, long-term studies using permanent plots measuring vegetation properties (e.g. height, diameter at breast height (DBH), or crown diameter, biomass and diversity indices) over time are commonly used to assess vegetation dynamics (Mugasha et al. 2017; Chidumayo 2019; ForestPlots.net et al. 2021; SEOSAW Partnership 2021). However, such studies are few within southern and central tropical Africa due to limited expertise, high costs, limited technological advancement, and logistical challenges of accessing and sampling large and remote geographical areas (Chidumayo 2019; SEOSAW Partnership 2021). Thus, exploration of environmental and land use gradients provides an alternative approach (Williams et al 2008; Syampungani et al. 2016) to understanding the resilience of Miombo woodlands to human disturbance. In our study we used remote sensing technology to understand land cover and carbon stock dynamics in Miombo woodland along a human disturbance gradient from the national park through the buffer zone to the communal area.

Field-based methods have long been used for aboveground biomass studies in Zimbabwe's Miombo woodlands focusing on developing biomass models (Frost 1990; Mushove 1994; Grundy 1995). While field data provide a primary source of AGC estimates that are important for national reporting under the United Nations Framework Convention on Climate Change (UNFCCC) and carbon projects such as REDD+, it has its challenges that include inaccessibility of field sample sites and the high cost of data collection making it unfeasible at times (McRoberts et al. 2014; Næsset et al. 2016). Novel solutions to some of the field data sampling challenges are being addressed by remote sensing (RS) technologies through increased precision of inventory estimates and reduced costs of forest resource inventory and monitoring at landscape scales (McRoberts et al. 2014; Næsset et al. 2016; Esteban et al. 2020). Thus, remote sensing products, including optical (e.g., Landsat, Sentinel 2A and LiDAR), synthetic aperture radar (SAR, e. g., Sentinel 1) data or their combination are now commonly used in vegetation assessment and monitoring (Ribeiro et al. 2008b; Saatchi et al. 2011; Mitchard et al. 2011; Harris et al. 2012; Vibrans et al. 2013; Hansen et al. 2015; Macave et al. 2022). Optical images are dependent on atmospheric conditions at the time of data acquisition (Lu et al. 2016). However, SAR are active sensors that emit radiation at wavelengths that are less susceptible to atmospheric backscattering and thus have high transmissivity through clouds (Lu et al 2016; Urbazaev et al. 2018). The relationship between both types of remote sensing data (optical and SAR) and field data is used to develop models that can predict AGB at the landscape level (McNicol et al. 2018; Macave et al. 2022). However, both optical and SAR are affected by data saturation at high AGB (greater than 80 Mg ha^{-1} (Ribeiro et al. 2008b; Lu et al 2016; Urbazaev et al. 2018).

Multi-temporal L-band (23 cm wavelength) radar imagery has proven to be effective in detecting aboveground carbon and forest cover in woodland ecosystems including the miombo (Ryan et al. 2011; Joshi et al. 2017; Mitchell et al. 2017; Macave et al. 2022). L-band normalised radar backscatter (γ^0) can be used to model woody biomass (associated with its ability to penetrate the forest canopy) up to around 50 Mg C ha^{-1} (Ryan et al. 2011). For example, work in the region has shown that γ^0 has a strong correlation (r^2 0.61–0.76, p < 0. 0001) to biomass across several African landscapes (Mitchard et al. 2009; Ryan et al. 2011; McNicol et al. 2018). While advances have been made in the region in using remote sensing techniques to estimate AGB (Ribeiro et al. 2008b; Ryan et al. 2011; McNicol et al. 2018; Macave et al. 2022), in Zimbabwe there have been some studies that applied optical and SAR imagery (Gara et al. 2016; Dube et al 2018) and the Phased Array L-band Synthetic Aperture Radar is yet to be applied. Given the potential of remote sensing in estimating large-scale AGB, it has an important role in nature-based climate change mitigation projects, including REDD+, which necessitates exploration of its use in the country (Næsset et al. 2016). Thus, in this study we used L-band backscatter to assess the effects of human disturbance on aboveground woody carbon stocks and cover of Miombo woodland.

This study aimed to determine how woodland cover and AGC varied along a utilisation gradient at the interface of Mana Pools National Park and Chundu communal lands in north-east Zimbabwe. The specific objectives of the study were: (a) to map woodland cover, aboveground biomass and their changes between 2007 and 2017; (b) to quantify aboveground woody carbon stocks along the utilisation gradient over a 10-year period‘ and (c) to assess the utility of L-band radar for operational forest monitoring in Zimbabwe. We hypothesised that miombo AGC and their changes vary significantly along a utilisation gradient over time.

Methods and materials

Study area

The research was carried out at the interface of the Chundu communal lands, Ward 8 of Hurungwe district, and Mana Pools National Park, a protected wildlife area about 260 km west of Harare (Figure 1). The study area has a mean annual rainfall of 750 to 1 000 mm, concentrated between mid-November and the end of March (Anderson et al. 1993). Soils vary from loamy sand to sandy clay loam soils under typical Miombo woodland vegetation dominated by *Brachystegia* and *Julbernardia* species (Chivuraise et al. 2016). Subsistence agriculture is the primary source of livelihood, with maize, groundnuts, cotton and tobacco as the major crops, and cattle, goats and sheep as the major livestock (Ncube 2011).

The area represents a gradient from the national park where there is no formal access to forest resources, to the communal area with complete but controlled resource access. The study area was divided into three land use types following Muposhi et al. (2016): a national park (within the park boundary), a buffer zone (2–5 km from the park boundary) and a communal area (5 km from the park boundary) (Figure 1). The national park zone is 9 802.2 ha, the buffer zone 11 741.4 ha, and the communal area 9 798.6 ha.

The three land use types demonstrate a gradient in access of woodland resources given their different management regimes. Mana Pools National Park, which has been a UNESCO World Heritage Site since 1984 and core of the Middle Zambezi Biosphere Reserve, is managed by the Zimbabwe Parks and Wildlife Authority. Its resource management plan for fauna and flora includes early season burning to avoid severe late season fires and management of wildlife populations (Matsa et al. 2022). The buffer zone is predominantly a wildlife woodland area managed by the Rural District Council (RDC) on behalf of the communities.

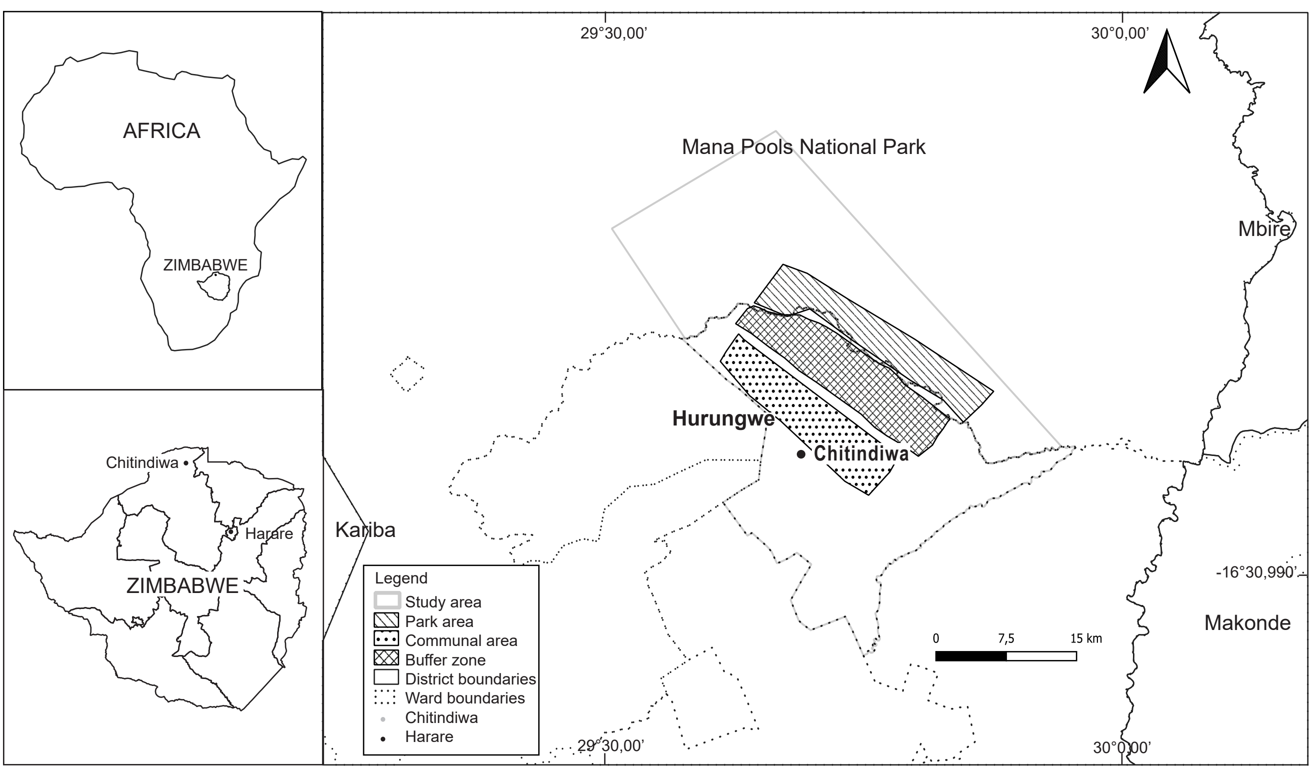

Figure 1: Location of the study area

The Council, however, has no clear management plan for this area except its reservation as wilderness area. In the past, RDC used the area for hunting concessions issued through the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) (Frost and Bond 2008). Plans are being revived for these activities. Population growth, mostly as a consequence of migration from the southern parts of Zimbabwe, resulted in an increase in communal settlements that now extend beyond the 5 km park buffer zone to about 1 km from the park boundary. Residents of the buffer zone are considered illegal settlers by local authorities. The communal area comprises several land use types that support the livelihoods of the communities, including crop lands, pasture lands and settlements. Most of the inhabitants of this area were resettled by the Government of Rhodesia from the Zambezi valley for the creation of Mana Pools National Park. The first inhabitants were resettled around the Chitindiva area (Figure 1) in the 1970s, some 20 km from the park boundary (Dzingirai and Mangwanya 2015). The population of Chundu grew from over 15 388 people and more than 3 293 households in 2012 to almost 18 765 people in nearly 4 198 households in 2022, with an average household of 4.5 people (ZimStat 2022).

In a related study Gotore et al. (2020) found anthropogenic disturbance in the study area to significantly differ with land use type for tree cutting (number of stumps) and observed fire counts (fire frequency per year). However, these differences in disturbance did not have a significant impact on the species composition and structure of the Miombo woodlands. The study presented here evaluated the impact of anthropogenic disturbance on aboveground biomass across the three contrasting land use types.

L-band processing, woodland cover, and aboveground carbon stock data

A combination of wall-to-wall mapping and random sampling approaches was used to assess AGC and cover of Miombo woodlands. Aboveground carbon stock maps of 2007 and 2017 were generated from 25 m horizontal send vertical receive (HV) polarisation pre- processed mosaic product of 2007 and 2017 radar backscatter images of the Phased Array L-band Synthetic Aperture Radar sensor on-board the JAXA Land Observation Satellite (ALOS-PALSAR 1 and 2 respectively). The mosaic product has terrain and radiometric corrections applied (McNicol et al. 2018). The radar data obtained from Shimada and Ohtaki (2010) were used to estimate aboveground carbon stock and woodland cover assessments following the methods presented by McNicol et al. (2018).

Calibration of the PALSAR mosaic involved converting integer values to units of radar backscatter (decibels) using the raster calculator tool in QGIS. This was done by applying the following equation (Shimada and Ohtaki 2010):

$$\gamma^0 = 10\log_{10} DN^2 - 83.0 \tag{1}$$

Where γ^0 is the backscatter in decibels and DN is the image in integer values.

The image in decibels was further converted to natural units to provide for arithmetic and not geometric means in subsequent analyses (Ryan et al. 2012) using the following formula:

$$\beta^0 = 10(\frac{\gamma^0}{10}) \tag{2}$$

Where β^0 is the backscatter image in natural units and γ^0 is the backscatter image in decibels.

Systematic differences were observed in the level of backscatter observed by ALOS-1 and ALOS-2 even where tree cover remained stable, perhaps because of differences in acquisition geometry and sensor characteristics. A correction factor was developed, based on a comparison of pseudo-stable locations from 2007 (ALOS-1) and 2017 (ALOS-2). A regular grid of points was generated across Southern Africa (every 0.5 degrees; n = 1 416), removing points with observations of forest cover loss (Hansen et al. 2013), on steep slopes (Farr et al. 2007) or on wetlands (European Space Agency GlobCover 2009 Project, http://dup.esrin.esa.int/page_globcover.php) to maximise consistency between measurements (remaining n = 1 001). ALOS-1 and 2 HV backscatter were extracted and compared using orthogonal regression (i.e., assuming errors on both axes). This provided a model to adjust the backscatter from ALOS-2 to match that expected from ALOS-1 (Figure 2). The resulting model (RMSE = 0.0207) was applied to the 2017 image in natural units.

$$\beta^{0^*}{}_{A2} = (0.6559 \times \beta^0{}_{A2}) + 0.00345 \tag{3}$$

Where $\beta^0{}_{A2}$ is the adjusted 2017 backscatter image in natural units and $\beta^0{}_{A2}$ is the 2017 backscatter image before correction in natural units.

Finally, AGC maps were generated by applying the following regional general model (r^2 = 0.57; cross validation RMSE = 8.5 Mg C ha^{-1}; bias = 1.1 Mg C ha^{-1}) of McNicol et al. (2018) which was developed in Miombo woodlands of Mozambique, Tanzania and Malawi for ALOS PALSAR 1:

$$AGC = 715.65 \times \beta^0 - 5.97 \tag{4}$$

Where AGC is aboveground carbon in Mg C ha^{-1} and β^0 is HV backscatter in natural units.

The Miombo woodlands of Mozambique, Tanzania and Malawi are similar to dry Miombo woodland (MAP 1 000 mm) found in Zimbabwe dominated by species of genera *Brachystegia* and *Julbernardia* (Gotore et al. 2020; Ribeiro et al. 2021a). Proportional random sampling points were established in each land use (national park = 94, buffer zone = 112 and communal area = 94) in a GIS environment from which aboveground carbon stock data was extracted from both the 2007 and 2017 maps using the point sampling plugin in Quantum GIS (QGIS) version 3.0.0.

Woodland cover for both 2007 and 2017 was based on the mapped AGC stock with reference to a woodland definition of > 10 Mg C ha^{-1} per pixel which was observed to be more or less consistent with other international forest definitions (McNicol et al. 2018). Zimbabwe's working definition for woodland comprises an area with trees with a minimum height of 5 m and a minimum canopy cover of 20% (Kwesha and Dreiser 1998). This definition aligns well with the Food and Agriculture Organization of the United Nations (FAO) definition of forest under Forest Resources Assessment Reporting (FAO 2020). The raster calculator tool in QGIS version 3.0.0 was used to calculate the woodland cover, loss and gains between 2007 and 2017.

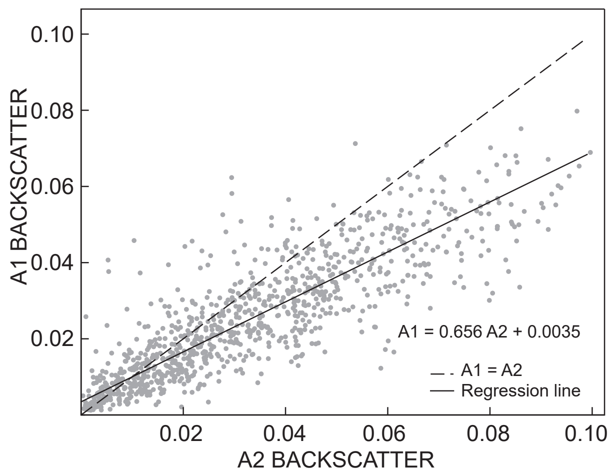

Figure 2: The relationship observed between backscatter from ALOS 1 and ALOS 2, based on pseudo-stable locations. A 1:1 relationship is shown with a dashed line, and the model used to correct ALOS 2 backscatter is shown by a solid line

Woodland area loss and gain were defined as area lost/gained divided by the wooded area in 2007.

Statistical analysis

A Kolmogorov–Smirnov test for normality was conducted for the AGC data (2007, 2017 and change) and only the AGC change data had normally distributed residuals ($p = 0.301$), while residuals for 2007 and 2017 AGC data were non-normal ($p < 0.05$). A one-way analysis of variance at a 95% confidence interval was conducted to test for differences in AGC change while a Kruskal–Wallis test was conducted to test differences in 2007 and 2017 AGC between the national park, buffer zone and communal area. All statistical analysis was conducted in R version 3.4.3 (R core Team 2017).

Results

Woodland cover

Figure 3 illustrates the spatial distribution of woodland cover in 2007 and woodland cover gains and losses by 2017 across land use types. Woodland cover for both 2007 and 2017 was higher in the buffer zone (42% and 43% respectively) and the national park (37% and 57% respectively) than in the communal area (27% and 31% respectively) (Table 1). Woodland loss between 2007 and 2017 was highest in the buffer zone (12%), followed by the communal area (9%), then the national park (7%) (Table 1). Conversely, the woodland cover gain was highest in the national park (23%) followed by the buffer zone (14%) then the communal area (14%) (Table 1). In all land use types, the woodland cover gain was higher than woodland loss over the 10-year period with a positive net change of 16% in the national park, 2% in the buffer zone and 5% in the communal area (Table 1).

Aboveground carbon stocks

The spatial distribution of aboveground carbon stocks (AGC) is illustrated in Figure 4. In 2007, mean AGC was not significantly different ($p > 0.05$) across land use types at 8.3 ± 0.6 Mg C ha−1 in the national park, 7.5 ± 0.7 Mg C ha−1 in the buffer zone and 6.1 ± 0.7 Mg C ha−1 in the communal area. In 2017 AGC had changed and was 17.1 ± 0.7 Mg C ha^{-1} in the national park, 12.0 ± 0.8 Mg C ha^{-1} in the buffer zone and 9.8 ± 0.95 Mg C ha^{-1} in the communal area, significantly lower in the buffer zone and communal area than the national park ($p < 0.0001$) (Table 2). Over the study period, there was a net gain in AGC in all land use types (Table 2). AGC gain was significantly higher ($p < 0.0001$) in the national park (8.9 ± 0.4 Mg C ha^{-1} per 10 years) than in the buffer zone (4.5 ± 0.5 Mg C ha^{-1} in 10 years) and communal area (3.8 ± 0.6 Mg C ha^{-1} in 10 years) (Table 2).

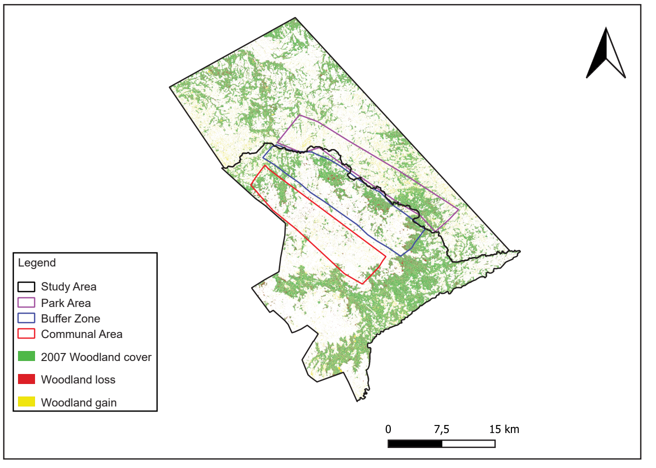

Figure 3: Woodland cover loss-and-gain map between 2007 and 2017 in the national park, buffer zone and communal area

Table 1: Percentage woodland cover, loss, gain and change in the national park, buffer zone and communal area for 2007 and 2017

Year	Land use type		
	National park	Buffer zone	Communal area
2007	37.0	42.4	27.1
2017	56.7	42.5	31.0
Loss	6.8	12.8	8.7
Gain	23.1	14.3	13.6
Net change	16.3	1.5	4.9

Discussion

Findings from the study fail to support the rejection of the hypothesis that aboveground carbon stocks (AGC) of miombo and their change vary significantly along a utilisation gradient over time. The dynamics of AGC differed among the three land use types. Over a 10-year period, AGC increased significantly in all land use types. The increase in woodland cover and AGC declined with elevated utilisation.

In 2017, woodland cover and AGC decreased with increased utilisation from the national park through the buffer zone into the communal area. Similar observations were made in Tanzania (Jew et al. 2016; Mganga et al. 2017) and Zambia (Sichone et al. 2019). Anthropogenic disturbance in the form of tree cutting, fire and vegetation clearing (Munishi et al. 2010; Gotore et al. 2020; Zinyowera et al. 2021), driven mostly by an expansion of tobacco fields and high demand for fuelwood for tobacco curing (Dzingirai and Mangwanya 2015; Chivuraise et al. 2016) resulted in decreased carbon stock. This observation collaborates earlier findings (Chidumayo 2013; Jew et al. 2016; Mganga et al. 2017). Our findings, however, are at variance with findings that recorded no direct relationship between disturbance and carbon stocks (Pelletier et al. 2017). These studies related carbon stocks to tree diversity (Pelletier et al. 2017; Amara et al. 2019). In general, studies in the region have shown that activities related to slash and burn agriculture, including subsistence use of Miombo woodlands, do not have much impact on carbon stocks as regeneration commonly offset carbon losses (Chidumayo 1990; Williams et al. 2008; Kalaba et al. 2013; McNicol et al. 2018).

The study observed increasing woodland cover in utilised areas between 2007 and 2017. This observation is not consistent with findings in the Mafungautsi forest in the Midlands province of Zimbabwe that indicated decreased woodland cover outside the protected area (Mapedza et al. 2003). Neither does it collaborate findings of a study in southern highlands of Tanzania where Miombo woodland cover increased with reduced forest utilisation (Lupala et al. 2015) and in Luanshya district of the Copperbelt province of Zambia where woodlands were shown to be generally declining in extent though with regrowth limiting this decline (Lembani et al. 2019). Several studies on land cover change in Zimbabwe attributed woodland cover loss to agriculture and settlement expansion (Kamusoko and Aniya 2007; Matavire et al. 2015; Nyelele et al. 2018). Findings from our study correspond to observations made in Matobo National Park, south-western Zimbabwe and surrounding areas, where forest area outside the national park increased by about 7% (Scharsich et al. 2017). This is attributed to regrowth in abandoned croplands (Scharsich et al. 2017). Further, since 2011 the study area has been placed under the Kariba REDD+ project (Dzingirai and Mangwanya 2015), which may have influenced the current increase in woodland cover. Most studies in the region, however, have demonstrated a general decline in forest cover over time (Kamusoko and Aniya 2007; Matavire et al. 2015; Kiruki et al. 2016; Mekonen et al. 2018), though this is offset by regrowth (McNicol et al. 2018).

AGC increased significantly in all land use types between 2007 and 2017. This points to the potential of Miombo woodland for carbon sequestration. Studies across the region have alluded to a positive potential of REDD+ in the miombo, with some pilot projects indicating positive results (Munishi et al. 2010; Lusambo and Lupala 2016; Lupala et

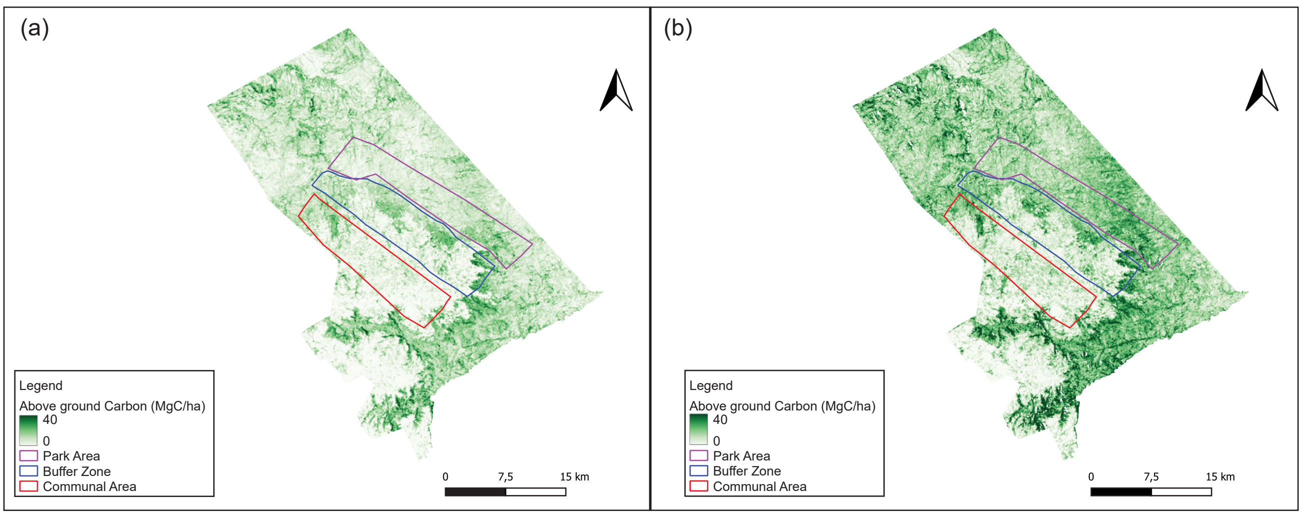

Figure 4: Spatial distribution of aboveground woody carbon stock density Mg C ha^{-1} in the national park, buffer zone and communal area in (a) 2007 and (b) 2017

Table 2: Mean aboveground carbon density (Mg C ha^{-1}) and carbon density change (Mg C ha^{-1} in ten years) in the national park (n = 94), buffer zone (n = 112) and communal area (n = 94) for 2007 and 2017

Year	Land use type			F value	*p* value
	National park	Buffer zone	Communal area		
2007	8.3 ± 0.6	7.5 ± 0.7	6.1 ± 0.7	2.84	0.060
2017	17.1 ± 0.7^{a}	11.97 ± 0.8^{b}	9.8 ± 0.95^{b}	18.84	0.0001*
2017–2007	8.8 ± 0.^{4} a	4.5 ± 0.^{5} b	3.8 ± 0.^{6} b	27.61	0.0001*

± = standard error
* = significant *p* value
Different superscript letters (a, b) following means within each row differ significantly (p < 0.05), based on Tukey's HSD test

al. 2017; Sichone et al. 2019). AGC estimates in the study area were within the range of estimates by Guy (1981) at Sengwa Wildlife Research Area (ranging between 12.58 t ha^{-1} –23.03 t ha^{-1}, between 1972 and 1976) and Dube et al (2018) at Mukuvisi woodlands (ranging from 7.4 to 56.1 Mg C ha^{-1}), but lower than estimates by Zimudzi and Chapano (2016) at Ngomakurira Mountain (34.5 to 65.1 t ha^{-1}) and a regional average of about 55 Mg ha^{-1} (27.5 Mg C ha^{-1}, Desanker et al. 1997). AGC estimates in miombo region are known to be variable depending on estimation method, sampling effort and land use history (Guy 1981; Desanker et al. 1997; Ryan et al. 2012; Ribeiro et al. 2013). Findings from our study, however, indicate that overall increases in AGC declined with increased utilisation, thus the need to consider anthropogenic disturbance as one major factor that has a negative impact on miombo carbon sequestration (Munishi et al. 2010). For example, future mitigation actions must therefore seriously consider anthropogenic factors by diversifying income sources, and market linkages and promoting sustainable utilisation, together, of course with climate change.

The observation of increasing AGC despite a growing population and ongoing disturbance in the study area is unusual. There are two possible interpretations of this finding: (i) that AGC is increasing across the three land use types; or (ii) that biases introduced by L-band data acquisition (e.g., soil moisture variation, data processing artefacts in the ALOS PALSAR mosaic product and residual differences in the characteristics of ALOS-1 and ALOS-2) are misinterpreted as widespread biomass increase. Distinguishing between these possibilities would require the collection of high-quality longitudinal reference data (e.g., through a field campaign, SEOSAW Partnership 2021) to validate the result of increasing biomass (Mitchard et al. 2009). Whilst there remains uncertainty in the overall increase in biomass that was observed, more confidence can be placed in the relative impacts between land use types. Even where there is a bias in one or both maps, the differences in losses/gains between areas would be expected to be robust, and therefore these results maintain relevance to assessing the relative impacts of woodland management regimes in Zimbabwe.

Carbon stocks in Miombo woodlands are not permanent as the miombo is a dynamic ecosystem with naturally varying amounts of tree cover and biomass over time. For example, the conversion of these woodlands to short-duration crop agriculture was projected to release large amounts of carbon dioxide into the atmosphere, as much as 50 Mg C ha^{-1} of AGC (Desanker et al. 1997). Though AGC in miombo are highly susceptible to anthropogenic and natural disturbance, studies have demonstrated that they can recover within 20 to 30 years (Williams et al. 2008; McNicol et al. 2015; Gonçalves et al. 2017). Further, studies globally have suggested that a widespread increase in biomass growth in African woodlands may be a result of elevated carbon dioxide, which in savannahs favour trees, especially in the more open, frequently burnt areas (Bond and Midgley 2012). However, other drivers including reduced burnt area, warmer and wetter climate and anthropogenic activity were observed to account for most of the biomass growth (Venter et al. 2018).

In general, our study demonstrated the potential for L-band radar imagery in woodland cover and biomass mapping. It also identified the need for high-quality reference data for calibration and validation of results. Woodland cover and AGC were both sensitive to land use types of protection, buffer zones and communal land, with the more restrictive land management practices generally associated with more woodland cover and higher average biomass. A general increase in both woodland cover and biomass over the past decade points towards the resilience of Miombo woodland ecosystems, their capacity to coexist with dynamic anthropogenic use, as well as their potential for climate change mitigation. These results provide the means to model the impact of management changes on woodland cover and carbon sequestration and point toward the value of monitoring biomass with L-band radar in African woodlands.

Conclusion

Aboveground carbon stocks vary along a utilisation gradient and have increased significantly over the past 10-year period in all land use types. These findings show that sustainable resource management is possible without the loss of such ecosystem services as carbon sequestration and climate change mitigation.

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