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Changes in forest cover and carbon stocks of the coastal scarp forests of the Wild Coast, South Africa

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Land-use intensification and declines in vegetative cover are considered pervasive threats to forests and biodiversity globally. The small extent and high biodiversity of indigenous forests in South Africa make them particularly important. Yet, relatively little is known about their rates of use and change. From analysis of past aerial photos we quantified rates of forest cover change in the Matiwane forests of the Wild Coast, South Africa, between 1942 and 2007, as well as quantified above- and belowground (to 0.5 m depth) carbon stocks based on a composite allometric equation derived for the area. Rates of forest conversion were spatially variable, with some areas showing no change and others more noticeable changes. Overall, the net reduction was 5.2% (0.08% p.a.) over the 65-year period. However, the rate of reduction has accelerated with time. Some of the reduction was balanced by natural reforestation into formerly cleared areas, but basal area, biomass and carbon stocks are still low in the reforested areas. The total carbon stock was highest in intact forests (311.7 ± 23.7 Mg C ha^{-1}), followed by degraded forests (73.5 ± 12.3 Mg C ha^{-1}) and least in regrowth forests (51.2 ± 6.2 Mg C ha^{-1}). The greatest contribution to total carbon stocks was soil carbon, contributing 54% in intact forests, and 78% and 68% in degraded and regrowth forests, respectively. The Matiwane forests store 4.78 Tg C, with 4.7 Tg C in intact forests, 0.06 Tg C in degraded forests and 0.02 Tg C in regrowth forests. The decrease in carbon stocks within the forests as a result of the conversion of the forest area to agricultural fields was 0.19 Tg C and approximately 0.0003 Tg C was released through harvesting of firewood and building timber.

Keywords: aboveground, basal area, belowground, biomass, soil carbon

Introduction

The Forest biome is the smallest biome in South Africa (Rutherford et al. 2006). The majority of indigenous forests occur in the Eastern Cape province, with a considerable proportion along the Wild Coast. There are approximately 50 000 ha of indigenous forest along the Wild Coast, largely made up of numerous fragments smaller than 100 ha within a grassland matrix (Rutherford et al. 2006; Berliner 2011). This forms part of the Maputaland-Pondoland-Albany biodiversity hotspot characterised by high species richness and endemism (CEPF 2011) and consequently there is much interest and responsibility to ensure that forests in the region are in good health and are used sustainably.

In common with savannas and forests elsewhere in the country, the Wild Coast forests have been, and continue to be, shaped by local people in pursuit of their livelihoods (Wigley et al. 2010; Shackleton et al. 2013). There are several human-mediated local-level pressures affecting the structure, composition and functioning of the forests. First is the clearance of forests to create arable fields and to a lesser extent built infrastructure. The second is the cutting and harvesting of timber and non-timber forest products from the forests to provide a wide array of livelihood needs (Timmermans 2004; Shackleton et al. 2007). Third is the extensive use of fire as a management tool in the surrounding grasslands (Kepe 2005), but which also may burn forest fringes and, if intense, at times burn into forest. Fourth, livestock are an important part of the local culture, identity and economy (de Klerk 2007) and their browsing and grazing pressures can influence forest structure and hinder recruitment of some forest species. Lastly, in the last few decades, several invasive alien species have proliferated in the region (Berliner 2011), with undoubted, albeit little studied, impacts on forest dynamics and composition (e.g. Jevon and Shackleton 2015).

Cognisant of these anthropogenic pressures and an increasing human population, the health and sustainability of the Wild Coast forests have been debated for some time. Rates of loss through clearing of forest to arable fields have, on the one hand, been mooted as excessive, certainly in the past, but on the other hand large areas of forest regrowth have been reported by the likes of Chalmers and Fabricius (2007), de Klerk (2007) and Shackleton et al. (2013). Some have argued that such regrowth is largely acacia invasion into grasslands and that it rarely progresses towards a greater species mix and biomass in comparison to indigenous forest (e.g. Berliner 2011). However, Shackleton et al. (2013) recently dispelled this argument, showing accumulation rates of approximately three woody species per decade after abandonment of former arable fields and basal area increments averaging

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0.34 m^2 decade^{-1}. The rate of forest succession depends upon the duration of the previous cultivation and once the field has been abandoned, the frequency and intensity of disturbances such as fire, harvesting and browsing.

Debate about the extent of forest reduction in this region have been largely at the local scale and driven by concerns about biodiversity loss. Internationally, interest in forest loss (or gain) is driven also by climate change concerns regarding the gains or loss of carbon through different land uses and land-use changes. Given their high biomass per unit area, forests store a considerable proportion of the world's terrestrial carbon (Backéus et al. 2005), and significant changes in forest biomass can affect atmospheric carbon levels. For example, during the period 2000–2005 approximately 4 million ha of forest was transformed annually in sub-Saharan Africa, with the conversion of forests to agricultural lands being the main contributor (approximately 59%) to such loss (FAO 2007). This forest conversion resulted in 0.29 Pg C y^{-1} of carbon emissions from Africa alone (Shvidenko 2008). Although South Africa is a signatory to the United Nations Framework Convention on Climate Change (UNFCCC), the focus of national climate change mitigation response measures has largely been within the energy and industrial sectors rather than the land-use sector (Rahlao et al. 2012), due to the small size of the woody biomes, nationally low rates of deforestation, and land tenure complexities. Nonetheless, Rahlao et al. (2012) argue that there is considerable scope for South Africa to participate in REDD+ (reducing emissions from deforestation and degradation). However, initial steps require detailed forest inventories of carbon stocks, as well as mapping of degraded areas for potential restoration. As yet, there are relatively few assessments of above- and belowground carbon in South African wooded biomes, such as thicket (e.g. Mills et al. 2005; Stickler and Shackleton 2015), savanna (Scholes and Walker 1993) and indigenous forests (Glenday 2007).

Within this context, this study sought to (1) determine the rate of forest reduction or gain in the central section of the Wild Coast and (2) translate the changes in forest abundance (areas and biomass) to changes in terrestrial carbon stocks.

Study area

The study was conducted in the Matiwane area of the Eastern Cape province, South Africa. The area extends from the Mbhashe River (32°35′94″ S, 28°81′39″ E) in the south to the Umzimvubu River (31°57′51″ S, 29°53′28″ E) in the north, a distance of 90 km and approximately 10 km in width from the coast, therefore totaling about 900 km^2. The mean annual rainfall is approximately 1 200 mm in the northern section and 950 mm in the southern section, with about 60% occurring during summer (October–May). The average annual evaporation rate is <1 400 mm (AGIS 2007) with summer temperatures ranging from 21 °C to 25 °C and winter temperatures ranging from 8 °C to 21 °C (AGIS 2007).

The geology is characterised by Table Mountain sandstone with a rocky coastal strip of Ecca sediments, mainly shales, along with intrusions of Karoo dolerite in the northern part (AGIS 2007). The southern section is made up of horizontally orientated Ecca Group shales, mudstones and sandstones and the Beaufort Group (grey fine-grained sandstone and mudstone) forming part of the Karoo sequence (AGIS 2007). Dolerite sheets and aquifers also occur in the area (AGIS 2007). The soils are mainly sandy loams with shallow clay soils occurring on eroded, rounded hills and spurs (AGIS 2007).

The area falls within the Indian Ocean Coastal belt of the Maputaland-Pondoland-Albany biodiversity hotspot (Mucina and Rutherford 2006). It is characterised by a fine-scale mosaic of grasslands and forests. The majority of the forest patches are less than 10 ha and are confined to valleys between rolling, grass-covered hills. However, there are a few patches of up to 200 ha or more (von Maltitz et al. 2003; Mucina and Rutherford 2006), including the Hluleka, Cwebe, Mpame and Pagela forests. According to Berliner and Desmet (2007) this region harbours critical biodiversity. The forests in this area are classified as vulnerable, with some forests near Port St Johns classified as endangered (Berliner and Desmet 2007). Coastal scarp forests in this region are comprised of low (up to 9 m) and medium-statured (15–25 m), species-rich forests with *Milettia grandis*, *M. sutherlandii*, *Buxus macowanii*, *B. natalensis* and *Umtiza listeriana* as common canopy species (von Maltitz et al. 2003). Beneath the canopy the forest is relatively open with mostly single-stemmed trees and a poorly developed herbaceous layer (von Maltitz et al. 2003). The forests are found on sloping coastal platforms and steep scarps in deep incised valleys at altitudes ranging from 0 to 600–800 m (von Maltitz et al. 2003).

The region is regarded as one of the most underdeveloped in South Africa. The majority of the population does not have access to piped water, sanitation or electricity. Poverty levels are high (de Klerk 2007). Key livelihood activities are small-scale agriculture and collection of non-timber forest products (Shackleton et al. 2007), which some external commentators deem to be detrimental to forest extent and quality (Obiri et al. 2002; Cawe and Geldenhuys 2007). Cash incomes come largely from government grants and migrant labour. Fires are common as people burn the grasslands to stimulate a flush of palatable growth in late winter for their livestock. This has also been implicated in forest fragmentation (von Maltitz et al. 2003; Cawe and Geldenhuys 2007).

Methods

Assessing forest cover change

Forest cover change was determined through mapping and assessment of the extent of forest cover on aerial photographs and satellite images from 1942 to 2007. Aerial photographs dated 1942, 1974 and 1995 were obtained from the National Geo-spatial Information Service (NGIS) and were georeferenced using ARGIS 9.2 (ESRI 2006) and rectified images (also obtained from NGIS) for the year 2003 were used as reference images for the georeferencing of the older images. Images referred to as 1942 were a combination of images obtained for 1938 and 1942 as the process of image photography was started in 1938 but only finished in 1942 for this area. After georeferencing, the images were overlaid to assess any change in forest cover over the years. Given that the old aerial images were georeferenced, all the area estimates were made on the 2007 SPOT 5 (Centre National d'Etudes Spatiales) images to avoid errors. All forest areas were then mapped through time, allowing identification on the final images of three classes, namely (1) intact forests (high forest cover and little or no change through time), (2) degraded forests (fragmented forest cover and decrease through time) and (3) regrowth forests (areas of grassland that have increasing tree cover through time). Mapped areas were ground-truthed during the field sampling through visits to any sites where image clarity was insufficient or classification ambiguous.

Forest structure and composition

Using GIS, 68 randomly located sample plots were distributed across the three mapped categories (32 in intact, 24 in degraded and 12 in regrowth forests). Each plot was 200 m² (40 m × 5 m), with the long axis running parallel to the contour. Within each plot the identity, diameter at breast height (1.3 m above ground level; DBH) and number of cut stems were recorded.

Allometry

General allometric equation for hardwoods and softwoods were developed across the 10 species that contributed most to the relative basal area of the forests. This was done via destructive felling and subsequent drying of 37 trees across a range of size classes up to approximately 60 cm DBH, which therefore covered approximately 92% of all the stems enumerated in the density plots. The following species and numbers were felled: *Cassipourea gerrardii* (1), *Celtis africana* (2), *Cussonia* sp. (7), *Englerophytum natalense* (4), *Ficus natalensis* (1), *Heywoodia lucens* (7), *Milletia grandis* (6), *Milletia sutherlandii* (5), *Strychnos henningsii* (2) and *Vepris lanceolata* (2). Each tree was felled at ground level after the DBH had been recorded. After felling, each was separated into main stem, branches and twigs, and leaves as per Brown (1997) recommendations. The fresh mass of each of the components was determined separately. A subsample of each was taken, weighed, dried for two weeks at 70 °C and reweighed to allow conversion to dry mass.

The oven-dry biomass of all trees were regressed against the DBH to obtain an allometric equation for the determination of aboveground dry biomass. The selection of the best form of the equation was made from power, linear, exponential and logarithmic functions of the dry biomass against DBH. The performance of each form of the equation was tested using the mean relative difference (RD) between the actual and the predicted dry biomass values (Zianis and Mencuccini 2004; Pilli et al. 2006). The RD is given by:

$$\mathrm{RD} = \left| B_p - B_a \right| / B_a \qquad (1)$$

where B_p and B_a denote the predicted dry biomass and actual biomass, respectively (Zianis and Mencuccini 2004; Pilli et al. 2006).

The mean RD of the Matiwane equation was compared with the mean relative differences of three internationally used generalised equations and one local equation developed for South African woodlands. The first was the 'global model' (Equation 2) developed by Zianis (2008) using meta-data from different authors around the world

$$\mathrm{AGB} = 0.1464(D)^{2.3679} \qquad (2)$$

where AGB and D are aboveground biomass and DBH, respectively.

The second was the WBE model developed by West et al. (1999) (Equation 3):

$$\mathrm{AGB} = 0.1(D)^{2.67} \qquad (3)$$

The third was the UNFCCC (2009) recommended equation for forests in regions receiving mean annual rainfall of 900–1 500 mm y^{-1}, developed by Brown (1997):

$$\mathrm{AGB} = \exp\{-1.996 + 2.32*\ln(D)\} \qquad (4)$$

The last was the allometric equation developed by Netshiluvi and Scholes (2001) for South African woodlands using meta-analysis of all available allometric equations from various studies

$$\mathrm{AGB} = 0.035(D)^{2.5} \qquad (5)$$

Carbon stocks

Aboveground carbon stocks

Using the derived allometric equation, the DBH measurement in each of the forest inventory plots was used to derive aboveground woody biomass per stem, which was summed for all stems in each plot. This was then multiplied by 0.5 to convert to aboveground woody carbon (IPCC 1996). For acacia species an allometric equation determined by Netshiluvi and Scholes (2001) was used for determining the aboveground biomass (Equation 6):

$$\mathrm{AGB} = 0.04\mathrm{DBH}^{2.6} \qquad (6)$$

Within each of the 40 m × 5 m forest inventory plots, four 1 m × 1 m quadrats were delineated at the corners for the estimation of carbon in litter, grass and the herbaceous layer. All litter present in each 1 m × 1 m quadrat was collected, bagged, oven-dried at 70 °C for two weeks and weighed. Litter was regarded as any detached dead organic plant material (fruit, flowers, leaves, twigs and small branches that were less than 10 cm in diameter). For larger deadwood litter (>10 cm diameter), the diameter of each piece was measured and the dry biomass determined as described for live trees, but with the estimate reduced by 10% to account for the loss of leaves, twigs and small branches (Delaney et al. 1998; Kirby and Potvin 2007). Carbon in the herbaceous layer was determined by clipping all forbs and grasses in the four 1 m × 1 m quadrats, the material was bagged and the samples oven-dried also at 70 °C for two weeks and weighed. Final dry weights were multiplied by 0.5 to provide carbon content (Ordónez et al. 2008).

To determine carbon removed as a result of recent harvesting, in each of the 40 m × 5 m plots the diameter of each cut stem was recorded, with the diameter taken at the point where the cut was made if below 1.3 m high. The derived allometric equation used to determine carbon in hardwoods was used to estimate the carbon removed through the harvest of these trees.

Belowground carbon stocks

Belowground root biomass was estimated using the regression equation developed by Cairns et al. (2003) which relates the aboveground biomass to root biomass (Equation 7). The belowground biomass values where converted to carbon by multiplying by 0.5.

$$BGB = \exp\{-1\ 085 + 0.926\ln(ABD)\qquad (7)$$

Soil carbon was estimated from cores taken from each of the four 1 m × 1 m quadrats. First, the soil surface was cleared of litter. A 15 cm diameter soil core was taken to 0.5 m depth; rocks and large roots prevented sampling deeper than this. Each soil core was divided into five depth classes (0–3, 3–5, 5–10, 10–20 and 20–50 cm). The soil samples were air-dried for 4 d and then thoroughly mixed between the four 1 m × 1 m quadrats. The roots were removed and the soil samples sieved using a 2 mm sieve, and then carbon content determined via complete combustion in an elemental analyser (with no separation of inorganic carbon, which is considered to be low in these soils). Separate cores were taken using a steel tube of known volume for the determination of soil bulk density. The soil carbon stocks were determined as the product of carbon concentration, bulk density, depth and area, and extrapolated to a per hectare basis.

The carbon stocks in each component were expressed on a per hectare basis and summed to provide total carbon stocks. The total carbon in each forest state was attained by multiplying the carbon stock of the forest class by the total area of the forest class. The difference in the carbon pools were tested between forest classes using Kruskal–Wallis analysis of variance using OriginPro 8 (OriginLab 2007).

Results

Forest cover change

The aerial imagery analysis revealed a net decline in total forest area between 1942 and 2007 of 791 ha. This equates to 5.15% of the 15 352 ha of forests, or 0.08% y^{-1} (Table 1). However, there appeared to be an acceleration in the rate of clearing through time. Between 1942 and 1974, about 0.03% of forest area was transformed. Corresponding figures for 1974 to 1995 and then from 1995 to 2007 were 2.01% and 3.11%, respectively. This equates to 0.003% y^{-1}. for the first period and 0.10% and 0.26% for the second and most recent periods, respectively.

The clearing was concentrated in the area between the Mthatha and Mgazana rivers in the Ngqeleni and Port St Johns forest estates. On the southern side in the Bomvane Forest no clearing was observed from the aerial photographs. The canopy cover was reduced from >70% in intact forest to less than 20% cover in degraded patches. The analysis also revealed that close to 471 ha of the area that was clearly grassland in 1942 had some tree cover by 2007, which we termed regrowth forests (although they could be regarded as new forests if in areas not previously forested prior to 1942, but there is no way of knowing that from the current analysis). Mean canopy cover in these areas was 10–20%. Thus, the net reduction in forest cover over the 65-year period was only 320 ha.

The 24 field plots in degraded patches fell in previous and now abandoned agricultural fields. The clearing occurred as the forest was converted to agricultural fields primarily occurring on gentle slopes and valley bottoms. The collection of firewood and timber appeared to have had a minimal impact on forest degradation as compared with the conversion of forest to agricultural fields as there were only 14.0 ± 6.8 cut stems ha^{-1} in intact forests. Figure 1 shows an example of how the forest patches appeared on the 1942 and 2007 images, illustrating a degraded patch of forest near Mankosi village with the agricultural fields clearly visible from the image.

Forest composition

The conversion of the forest to agricultural fields resulted in a decline in the number of woody species present (Table 2) from 11.0 + 0.6 per 200 m^2 plot in intact forests to 1.0 ± 0.2 woody species in degraded forests ($p < 0.05$). The total number of recorded woody plant species in the intact forest plots was 82 compared with only two for the degraded forests and only one (*Vachellia karroo*, syn. *Acacia karroo*) in the regrowth forest plots. The most frequent species found in the cleared or degraded patches was *Heywoodia lucens*, occurring between the fields. Stem density was also markedly reduced in the degraded forests ($p < 0.005$), with intact forests having 1 840 ± 143.4 stems ha^{-1} and degraded forests 223 + 51.6 stems ha^{-1}. The stem density in regrowth forest was intermediate between these two.

Although 82 species were recorded in the intact forests, the basal area was dominated by relatively few species. *Heywoodia lucens* contributed over one-third to the total basal area, and the 20 most abundant species contributed 88.6% to the basal area of the Matiwane forests (Table 3).

Allometry

For the nine hardwood species a power function (Figure 2) gave the best results ($r^2 = 0.962$) compared with the other three functions (exponential $r^2 = 0.082$; linear $r^2 = 0.804$;

Table 1: Change in aerial cover of the Matiwane forests between 1942 and 2007

Period	Number of years	Forest decline		Forest gain	
		Total area (ha)	Annual decline (ha y^{-1})	Total area (ha)	Annual decline (ha y^{-1})
1942–1974	32	5.1	0.2	0	0
1974–1995	21	309.0	14.7	138	6.5
1995–2007	12	477.0	39.8	333	27.8

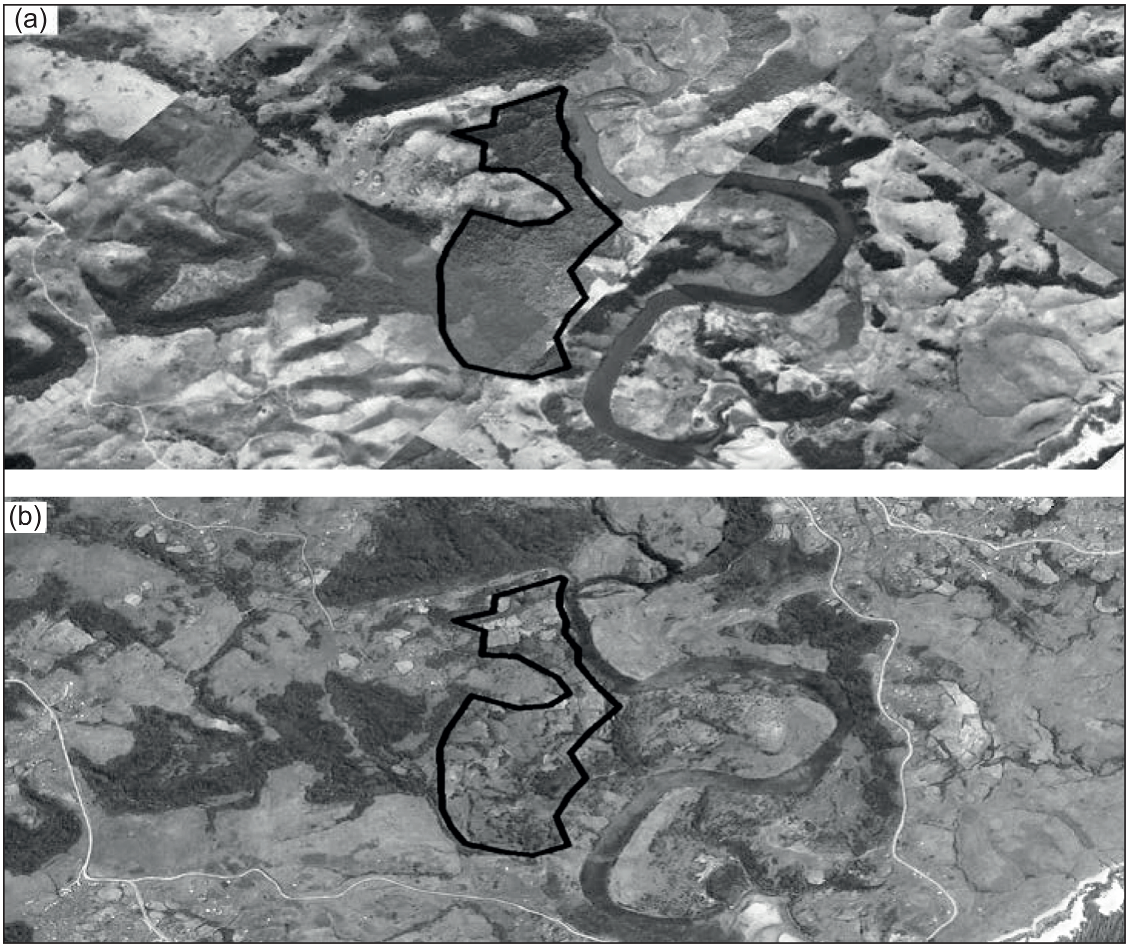

Figure 1: An example of forest conversion through comparison of the Mankosi Forest patch in 1942 (aerial photograph) and in 2007 (SPOT 5 image)

Table 2: Mean (±SE) density, basal area and species richness of forests in different states in the Matiwane area

Attribute	Forest state			*p*-value
	Intact	Degraded	Regrowth	
Number of woody species (200 m^2)	11.0 ± 0.57	1.0 ± 0.23	1.0 ± 0.08	<0.05
Stem density (stems ha^{-1})	1 819 ± 141	223 ± 51.6	1 550 ± 361	<0.05
Tree density (trees ha^{-1})	1 435 ± 79	162 ± 39.4	900 ± 172	<0.05
Basal area (m^2 ha^{-1})	43.3 ± 4.8	10.7 ± 4.8	5.6 ± 0.8	<0.05

Table 3: Species contributing 1% or more to the mean basal area of the Matiwane forests

Species	Relative basal area contribution (%)	Species	Relative basal area contribution (%)
Heywoodia lucens	36.4	*Maytenus acuminata*	1.7
Vepris lanceolata	6.9	*Dalbergia multijuga*	1.7
Cussonia sp.	6.0	*Rothmania capensis*	1.7
Strychnos henningsii	4.8	*Chaetachme aristata*	1.6
Milletia grandis	4.7	*Combretum* sp.	1.6
Ficus natalensis	4.4	*Lantana camara*	1.6
Celtis africana	3.1	Unknown (sabhokwe)	1.3
Millletia sutherlandii	2.7	Unknown (umbotyane)	1.0
Cassipourea gerrardii	2.6	*Pavetta lanceolata*	1.5
Englerophytum natalense	2.3	*Trichilia dregeana*	1.0

log $r^2 = 0.591$). In addition, it gave the lowest mean relative difference value (0.34) compared to the other three forms examined (exponential RD = 1.3; linear RD = 14.6 and log RD = 40.4). All four versions of the softwood equation gave strong correlations ($r^2 > 0.8$). The power function provided the best fit ($r^2 = 0.95$) and a mean relative difference of 0.32 ± 0.43 (Figure 2).

The best-fitting equations for the hardwoods for the relationship between DBH and total leaf biomass, stem dry biomass or branch dry biomass, and between leaf dry biomass and total tree biomass are shown in Figure 3. Two functions, namely the linear and power functions, gave better predictions, with the linear function effective in relating DBH to total leaf biomass ($r^2 = 0.887$) and relating total leaf dry biomass to total tree biomass ($r^2 = 0.881$). The power function was effective in relating DBH to total stem dry biomass ($r^2 = 0.962$) and relating branch dry biomass ($r^2 = 0.933$) (Figure 3).

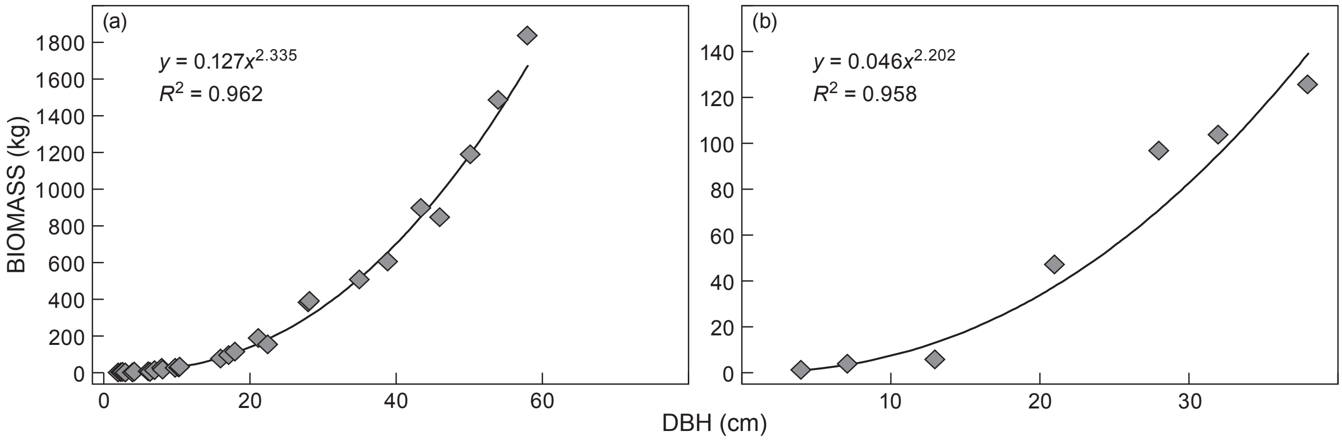

Figure 2: General allometric relationship for (a) hardwoods and (b) softwoods (*Cussonia* sp.) in Matiwane forests

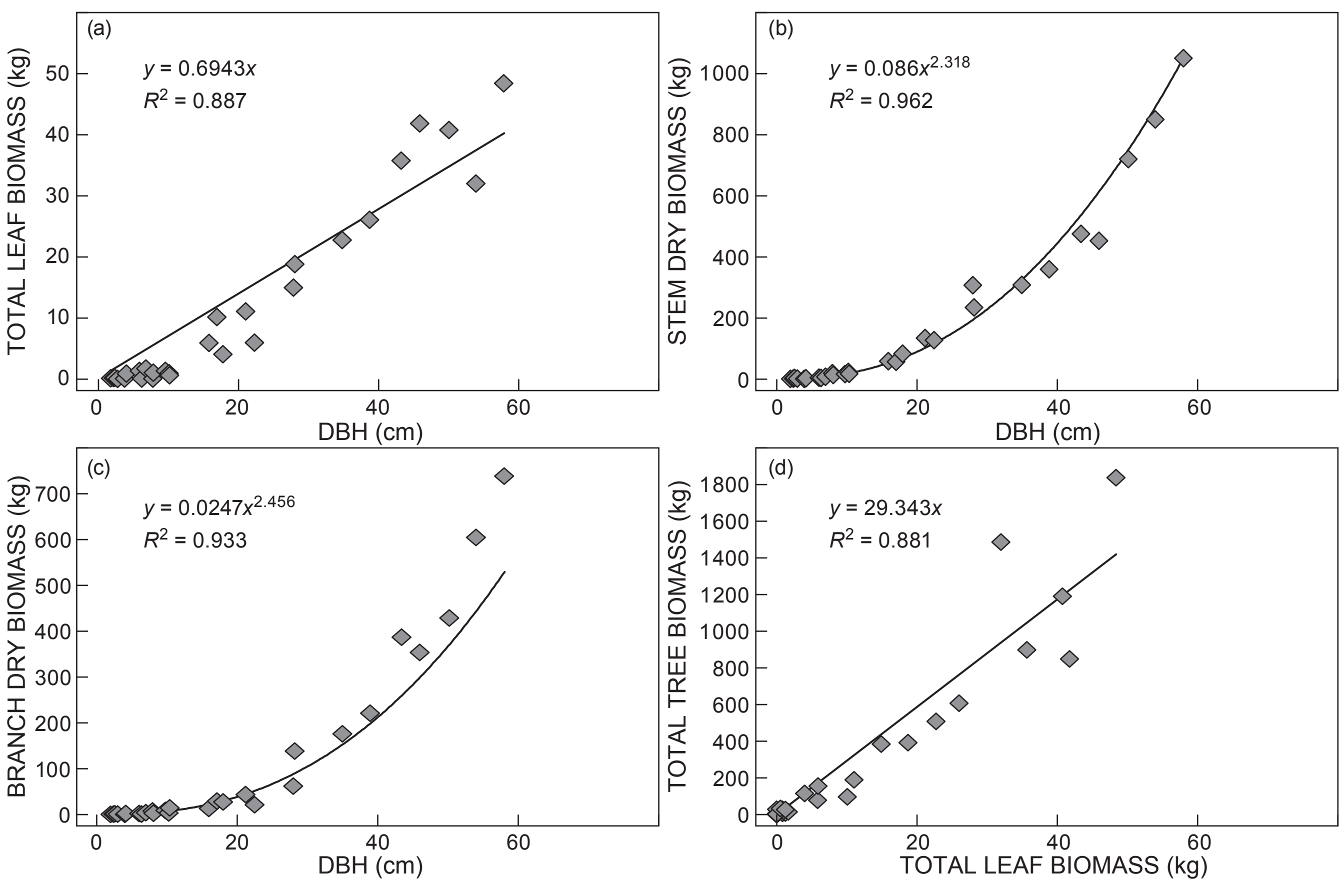

Figure 3: Allometric equations relating diameter at breast height (DBH) to different tree components for hardwoods

The five allometric equations obtained from the literature differed in estimating the total tree biomass in relation to the actual recorded data. Our equation demonstrated a good fit, with a mean RD of 0.14 ± 0.80 compared with the other four generalised equations. Brown's (1997) equation was the closest with a RD value of 0.17 ± 0.84. West's (1999) WBE equation had the largest discrepancy with a mean RD of 1.12 ± 1.11, i.e. greatly overestimating the dry biomass, whereas the Netshiluvhi and Scholes (2001) savanna equation underestimated the dry biomass (relative difference = 0.53 ± 0.28).

Carbon stocks

The total carbon stock varied amongst the three forest states (Table 4), with the intact forest having significantly higher total carbon stocks (311.7 ± 23.7 Mg C ha^{-1}) than the degraded (73.5 ± 12.3 Mg C ha^{-1}) and regrowth forest areas (51.2 ± 6.2 Mg C ha^{-1}). The greatest contribution to total carbon stocks was soil carbon (to a depth of 0.5 m) in all three forest types (Table 4), contributing 54% in intact forests, 78% in degraded forests and 68% in regrowth forests. Considering the aboveground stocks alone, over 87% was contained in woody plants greater than 1.3 m tall, being 87.6% for intact forests, 88.3% in degraded forests and 95.1% in regrowth forests. The stocks in the herbaceous and litter layers were relatively small. The intact forests had significantly higher ($p < 0.001$) carbon stocks than degraded or regrowth forests in all aboveground pools with the exception of the herbaceous layer.

The soil carbon content was highest in the upper soil layers and decreased with depth. In the intact forests the proportion of soil carbon in the upper layers (25% in the top 0.05 m and 43.9% was in the upper 0.1 m) was lower than that in the degraded (33.6% and 61.4%, respectively) and regrowth forests (34.7% and 63.3%, respectively). When comparing total carbon stocks in the degraded forests to the intact forest, it is evident that the degraded areas have less than approximately 75% of their original carbon pools.

The Matiwane forests store a total of 4.78 Tg C, with 4.7 Tg C in intact forests, 0.06 Tg C in degraded and 0.02 Tg C in regrowth forests (Table 4). The total estimated reduction in carbon stocks as a result of the conversion of the forest area to agricultural fields was 0.19 Tg C and approximately 0.0003 Tg C was removed through harvesting of firewood and building timber. Approximately 0.024 Tg C was accumulated in the regrowth forests.

Discussion

Forest cover change

Aerial photography and SPOT 5 imagery assessment revealed that there have been low levels, albeit continuous, clearing of forests in the region, largely concentrated in the area between the Mthatha and Mgazana rivers. The area south of Mthatha River in the Bomvane Forest Estate and the area north of Mgazi River towards Port St Johns displayed no noticeable forest cover change over the 65-year period. The concentration of the clearing in between the Mthatha and Mgazana rivers may be due to the difference in agricultural practices or the strength of local governance institutions. In the southern region near the Bomvane Forest Estate inhabitants mainly have small gardens close to their homesteads, whereas in the area between the Mthatha and Mgazana rivers the residents have larger fields some distance away from the homestead (Andrew 1992, cited in Fay 2009). This trend is reported to have been facilitated by local magistrates, who occasionally warned headmen about allowing local residents to extend their gardens beyond their homesteads (Fay 2003, 2009).

The results show an increase in the rate of forest conversion, being highest in the most recent period. Cawe (1986) describes that there was a significant increase in the rate in the late 1970s after the Transkei 'homeland' was granted nominal 'independence' from South Africa, but lacked the resources to control or manage the forests. The further increase in the rate post-1995 corresponds

Table 4: Mean (±SE) above- and belowground carbon pools in intact, degraded and regrowth forests in the Matiwane area

Carbon pool	Forest state						Significant p-value between forests
	Intact		Degraded		Regrowth		
	Mg C ha^{-1}	%	Mg C ha^{-1}	%	Mg C ha^{-1}	%	
Aboveground							
Trees	113.7 ± 14.9	36.5	13.6 ± 7.1	18.5	12.4 ± 5.0	24.2	<0.001
Litter	6.4 ± 0.6	2.1	0.5 ± 0.2	0.7	0.08 ± 0.02	0.2	<0.005
Large litter	9.7 ± 1. 5	3.1	0.6 ± 0.2	0.8	0.06 ± 0.4	0.1	<0.001
Herbaceous	0	0	0.7 ± 0.1	1.0	0.5 ± 0.03	1.0	>0.05
Belowground							
Tree roots	15.2 ± 1.9	4.9	1.9 ± 0.9	2.6	1.9 ± 0.7	3.7	<0.01
Total soil (0–0.5 m depth)	167.1 ± 4.9	53.6	57.2 ± 4.1	77.8	34.9 ± 7.1	68.2	<0.001
0–0.03 m	24.1 ± 0.6		11.4 ± 1.4		7.8 ± 0.6		>0.05
0.03–0.05 m	18.1 ± 0.5		7.8 ± 0.4		4.3 ± 0.4		>0.05
0.05–0.1 m	31.9 ± 0.9		15.9 ± 0.9		10.0 ± 1.2		>0.05
0.1–0.2 m	45.0 ± 3.1		22.1 ± 1.4		12.9 ± 4.9		>0.05
0.2–0.5 m	48.0 ± 19.5		28.6 ± 1.0		26.0 ± 1.8		>0.05
Total (Mg C ha^{-1})	311.7 ± 23.7		73.5 ± 12.3		51.2 ± 6.2		
Forest area (ha)	15 352		791		471		
Total carbon (Tg C)	4.70		0.06		0.02		

to another political transition, namely the reincorporation of the Transkei homeland back into South Africa after the democratic transition. This was accompanied by a further weakening of forest governance structures on the ground (von Maltitz and Shackleton 2004). A similar increase in conversation rates post the 1994 democratic transition was noted by Stickler and Shackleton (2015) for thickets on the Bathurst commonage. However, whilst these political changes were at a national scale, the rate of clearing did not accelerate throughout the Matiwane area, suggesting that local-level factors also play a role. For example, we found no clearing in the southern parts of the study area, nor in the northern section. This latter finding contradicts the assertion by Obiri and Lawes (2004) that the forests in the Port St Johns area had declined in extent during the previous 25 years. We found no noticeable clearing in the area north of Mgazana River during the 65-year period.

The analysis revealed that while the rate of forest conversion was low, but accelerating, there were also areas where woody plants were invading into what was earlier mapped as grassland, demonstrating a spatially and temporally dynamic mosaic of shifting forest patches and boundaries. These were estimated to cover 471 ha, representing approximately 60% of the total forest area transformed. The woody plant species richness of these regrowth sites was low, dominated by the pioneer species *V. karroo*, regarded as a precursor for forest development (Cawe and Geldenhuys 2007). Shackleton et al. (2013) recently showed that woody plant species richness increased steadily after abandonment of cultivated fields, gaining approximately three woody species per decade after abandonment. Basal area accumulation was slower at approximately 0.34 m^2 $decade^{-1}$. This may also correspond to the internationally noted phenomenon of woody plant encroachment into grasslands, potentially a result of increased atmospheric carbon dioxide levels along with altered fire and herbivory regimes (Bar et al. 2004; Hanberry and Hansen 2015). Given these dynamics it would be useful to assess changes over the more recent past, i.e. since 2007 to present.

Forest composition and structure

Both the aerial photographic analysis and the ground-truthing revealed that the main cause of forest decline was conversion to agricultural fields. Agriculture used to be the mainstay of local livelihoods, but it has declined significantly in the last few decades (Andrew and Fox 2004; Shackleton et al. 2013), following trends in communal areas elsewhere in the province (Hebinck and Lent 2007). This then contradicts that land clearance is the primary driver of the observed accelerating rate of forest decline. However, it may be that although fewer households are practicing agriculture, the farm size of those still working the land is increasing, resulting in continued forest reduction in some areas. Alternatively, the spatial variation observed in clearing of different parts of the forest also applies to agriculture, with some areas expanding, whilst other areas are declining.

Collection of firewood and poles for building are additional pressures that previous authors have reported as leading to forest decline. Unless intensive, these cannot be quantified by aerial photography. However, the density of cut stems in the forest plots was relatively low (14 ± 6.82 cut stems ha^{-1}) in intact forests. In addition, there was no difference in stem or tree density or basal area between intact forests in the communal areas and intact forests within protected areas. Furthermore, the mean basal area (43.3 ± 4.77 m^2 ha^{-1}) was comparable with that in other studies in the region and South African scarp forests in general (35–57 m^2 ha^{-1}) as reviewed by Lawes et al. (2004). These three measures suggest that the harvesting pressures are low and are unlikely to have resulted in significant deforestation.

Allometry

There are numerous forms of allometric equations for determining the aboveground biomass of forest trees, including the power, linear, logarithmic and exponential functions. As discussed by Pilli et al. (2006) our results indicated that the power function provided the best fit in estimating aboveground biomass. However, the power function was not effective in relating DBH to total leaf biomass and relating total leaf biomass to the total aboveground biomass in hardwoods, as compared with the linear function which gave the best fit. Of the five published allometric equations compared, the Matiwane and Brown (1997) equations gave the best estimates for the aboveground biomass with only 13% and 17% relative difference to the actual biomass.

The scaling exponent is usually a value of between 2 and 3 (Zianis and Mencuccini 2004), which was also the case with the derived allometric equation, being 2.335 for hardwoods (r^2 = 0.962) and 2.202 for softwoods (r^2 = 0.958). The data from this study indicated that the West et al. (1999) (herein referred to as the WBE equation) fractal model does not apply for the Matiwane forest, which proposed that the scaling exponent should equal 2.67 independent of species, site, structural and morphological characteristics of trees (Zianis and Mencuccini 2004; Pilli et al. 2006). The scaling exponent of the Matiwane equation is well below the value proposed by West et al. (1999). The WBE equation has been questioned, with some authors arguing that the use of a universal scaling exponent is unwise as the ratio of biomass to DBH for trees growing in different environments is not constant (Zianis and Mencuccini 2004; Pilli et al. 2006).

Carbon stocks

Given the significant reduction in the number of trees when forests are cleared, aboveground carbon stocks are reduced, as may be the belowground stocks. In this study the reduction in total carbon stocks (including belowground to 50 cm depth) was approximately 76%, from 311.7 ± 23.69 Mg C ha^{-1} in intact forest to 73.5 ± 12.34 Mg C ha^{-1} in the degraded areas. The regrowth forests had 51.2 ± 6.18 Mg C ha^{-1}. Similar magnitudes of reduction, greater than 200 Mg C ha^{-1}, have been recorded elsewhere, such as Cameroon (Kotto-Samme et al. 1997). However, direct comparison between studies is compromised due to differences in soil sampling depths when estimating belowground and soil carbon. The greatest differences in carbon pools among the three states were in the vegetation and in the soil, indicating the vulnerability of these two pools to the conversion of forest to arable fields. Considering the extent of forest and the rate of forest conversion, there has been a reduction by approximately 0.19 Tg C between 1942 to 2007, which is small (4%) relative to the current stock of 4.7 Tg C. Even though this assumes 1942 as the baseline, some clearing is likely to have occurred prior to that period. However, because human populations at that time were only a fraction of what they are today, it is unlikely that the extent of forest conversion prior to that period was larger.

Aboveground carbon

The conversion of the forest to agricultural fields reduced the total tree carbon density by 88% from 128.9 ± 16.7 Mg C ha^{-1} in intact forest to 15.6 ± 8.0 Mg C ha^{-1} in degraded forests. This echoes the findings of Hughes et al. (2000) who showed a 95% reduction at Los Tuxtlas Biological Station in Mexico subject to forest clearance, and also Kauffman et al. (1995) who showed a 58–112 Mg ha^{-1} reduction in aboveground carbon pools in Amazon forests. This clearly reflects on the susceptibility of this carbon pool to forest conversion. The regrowth forests had similar woody carbon stocks (14.4 ± 5.7 Mg C ha^{-1}) to the degraded forest. The carbon stock of the intact forests was well within the range reported from a number of tropical and temperate forests internationally (Brown et al. 1993; Lal 2005; Gibbs et al. 2007). Prentice (2001), cited by Lal (2005), presented a global mean of 60–130 Mg C ha^{-1} for temperate forests and 120–194 Mg C ha^{-1} for tropical forests. However, within South Africa, Glenday (2007) reported values of aboveground carbon of 99 Mg C ha^{-1} for scarp forests and 105 Mg C ha^{-1} for coastal forest in KwaZulu-Natal province, which is significantly lower than our results for Matiwane and international figures. However, there was not much difference between the estimates of belowground carbon density between our results (15.2 ± 1.8 Mg C ha^{-1}) and those of Glenday (2007) (18 ± 2 Mg C ha^{-1}).

In a similar manner to the decline in tree biomass and carbon stocks, there were marked reductions in carbon stocks in litter and larger deadwood between intact and degraded forests. The regrowth forests were relatively similar to the degraded forests in this regard. Litter carbon declined by 87% and deadwood carbon by 94%. This is due to decreased litter and deadwood fall as mature trees are removed, collection of deadwood for fuel and the burning of cleared areas (Lasco 2002). In contrast, there was a higher carbon stock in herbaceous plants in the degraded and regrowth forests than the intact ones. That is a consequence of the dense tree cover in intact forests suppressing the presence and growth of herbaceous plants.

Soil carbon (to 50 cm depth)

The conversion of forest lands to agricultural lands lowers soil carbon because of the reduction in litter input and leaching (Almendros et al. 2005; Lal 2005). Guo and Gifford (2002) revealed a 42% reduction in soil carbon stocks following the conversion of forests to cropland from their metadata analysis of 74 publications. Our results revealed a 66% reduction, from 167.1 ± 4.9 Mg C ha^{-1} in intact forests to 57.2 ± 4.1 Mg C ha^{-1} in degraded states.

The soil pool had a higher carbon content than any other pool in the forest, followed by living standing trees including both above- and belowground pools. The soil carbon pool contributed more than 50% towards the total carbon density of the three states, with 54% contribution in intact forest, 78% in degraded and 68% in regrowth forests. The soil carbon content declined with increasing depth as reported by others (e.g. Ordónez et al. 2008).

Conclusion

In conclusion, this study has shown that the extent of forests in the Matiwane area has been remarkably stable, albeit spatially dynamic, over the last several decades despite a pervasive narrative of forest loss and destruction by local people and injudicious use of fire. Not only has the extent of forest decline loss been low, but there is also evidence of formerly cleared areas reforesting via natural succession. However, the analysis indicates that the rate of decline, however small, has increased through time, but without clear reasons for such. This requires monitoring and more on-the-ground analysis.

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