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Unsustainable fuelwood extraction from South African savannas

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Abstract
Wood and charcoal supply the majority of sub-Saharan Africa's rural energy needs. The long-term supply of fuelwood is in jeopardy given high consumption rates. Using airborne light detection and ranging (LiDAR), we mapped and investigated savanna aboveground biomass across contrasting land uses, ranging from densely populated communal areas to highly protected areas in the Lowveld savannas of South Africa. We combined the LiDAR observations with socio-economic data, biomass production rates and fuelwood consumption rates in a supply–demand model to predict future fuelwood availability. LiDAR-based biomass maps revealed disturbance gradients around settlements up to 1.5 km, corresponding to the maximum distance walked to collect fuelwood. At current levels of fuelwood consumption (67% of households use fuelwood exclusively, with a 2% annual reduction), we calculate that biomass in the study area will be exhausted within thirteen years. We also show that it will require a 15% annual reduction in consumption for eight years to a level of 20% of households using fuelwood before the reduction in biomass appears to stabilize to sustainable levels. The severity of dwindling fuelwood reserves in African savannas underscores the importance of providing affordable energy for rural economic development.

Keywords: fuelwood, communal land, LiDAR, biomass, savannas, supply–demand

1. Introduction

Over 80% of households across sub-Saharan Africa rely on biomass as their primary energy source (IEA 2010), mostly for cooking and heating (Scholes and Biggs 2004). South Africa is similarly dependent on biomass as an energy source, despite being a comparatively well developed country with rapidly improving rural access to electricity (Shackleton and Shackleton 2004). Approximately 55% of the 2.4 million rural households across South Africa have access to electricity (Pereira *et al* 2011), yet 54% of rural households continue to use wood as their main source of energy (Serwadda-Luwaga

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and Shabalala 2002, Madubansi and Shackleton 2006). This situation is driven primarily by the costs of purchasing a stove and additional electricity. Approximately 712 800 rural households connected to the national electricity grid still cook with fuelwood provided by savannas. The value of this ecosystem service can be expressed as a national savings in electricity of 375 MW and savings on new electricity generation of 14.6–77.4 million USD per year for coal-fired and open cycle gas turbine power stations, respectively (appendix). Nationally, rural households use from 4.5 to 6.7 million tonnes of fuelwood per year (Shackleton and Shackleton 2004, Pereira *et al* 2011). The provision of biomass by the ecosystem is thus of considerable value, and sustainable management is therefore essential to energy security and poverty alleviation.

The dependence on fuelwood is especially high in the communal areas of the former 'homelands'—self-governing territories established through forced resettlement of 3.5 million people during the apartheid era (Shackleton *et al* 2001). Today these state-owned areas support high human population densities of 150–300 people km^{-2}, and unemployment levels are high (Pollard *et al* 2003). In the communal areas of the Lowveld of South Africa, the average household use 3–4 tonnes (t) of fuelwood per annum (Madubansi and Shackleton 2007, Banks *et al* 1996, Shackleton *et al* 1994). Almost a decade after the introduction of electricity, over 90% of households still used fuelwood for cooking and the mean household consumption rates has not reduced (Madubansi and Shackleton 2007). Strong dependence on fuelwood and resulting high levels of extraction has raised concerns about a looming 'fuelwood crisis' at local and national scales (Dovie *et al* 2002, von Maltitz and Scholes 1995, Williams and Shackleton 2002, Banks *et al* 1996, Higgins *et al* 1999). If harvested within sustainable limits, fuelwood could continue to supplement rural energy requirements and represent a valuable renewable energy source (Ghilardi *et al* 2009, Williams and Shackleton 2002). However, if used unsustainably, it can lead to continued impoverishment of rural communities and long-term woodland degradation. Sustainability is achieved when the long-term supply (wood standing crop plus recruitment and production) exceeds demand (harvesting and consumption) (Banks *et al* 1996). Since neither the biomass supply nor the harvesting pressure is uniformly distributed, spatially explicit data are required to model the system (Ghilardi *et al* 2009, Williams and Shackleton 2002, Banks *et al* 1996).

To quantify the impact of fuelwood extraction on patterns of aboveground woody biomass (hereafter referred to only as biomass) in savanna ecosystems, we used light detection and ranging LiDAR (Koch 2010, Colgan *et al* 2012, Asner 2009) from the Carnegie Airborne Observatory (CAO) to estimate biomass over 25 000 ha of sample sites in the Lowveld of South Africa (figure 1). Firstly, we quantified and compared the biomass of communal areas in the Lowveld to reference sites in neighbouring conservation areas, specifically the Kruger National Park (KNP) and the Sabi Sand Private Game Reserve (SSGR) (figure 1). These

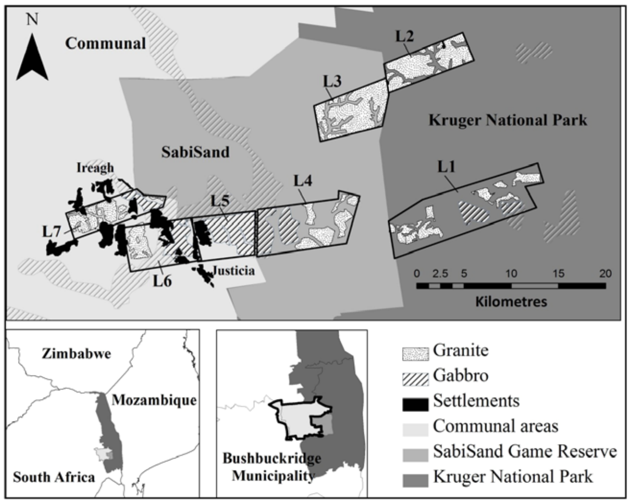

Figure 1. Study area in the Lowveld of South Africa, including (from east to west) Kruger National Park, Sabi Sand Game Reserve and communal areas in the Bushbuckridge municipality. Sites are delineated as L1–7. Gabbro intrusions are indicated within the predominantly granitic landscape. Human settlements are also mapped.

conservation areas exclude humans and domestic livestock, but the savanna vegetation remains affected by grass fires and large mammalian herbivores, notably elephants (Shannon *et al* 2008b, Wessels *et al* 2011, Smit *et al* 2010, Asner *et al* 2009, Sankaran *et al* 2005). Second, we also investigated biomass gradients previously observed around settlements in communal areas (Banks *et al* 1996, Shackleton *et al* 1994) to shed light on the prevalence of self-collection on foot and the accessibility of biomass as a resource. Fuelwood extraction is the major driver of such gradients (Banks *et al* 1996, Shackleton *et al* 1994) as the only other potential driver, namely, overgrazing by cattle, leads to bush encroachment rather than decreases in woody biomass (Scholes 2009). Finally, we combined LiDAR-derived biomass maps with ecological and socio-economic data in a supply-and-demand model (Banks *et al* 1996) to test the future sustainability of fuelwood extraction under various scenarios. Detailed settlement-specific information on the population and their fuelwood consumption has been collected during household surveys (Kahn *et al* 2007). The study therefore first investigated the differences in biomass between land uses (communal versus conservation), then biomass gradients around communal settlements, before finally focusing on one settlement and its secluded woodland to apply a supply–demand model to predict future trends in biomass resources.

Based on previous field studies, we hypothesized that, (i) biomass should be lower in communal areas compared to conservation areas due to fuelwood extraction, (ii) biomass gradients would be prevalent in woodlands around settlements (Fisher *et al* 2012, Banks *et al* 1996, Shackleton *et al* 1994) due to current and historic collection of fuelwood on foot and (iii) the biomass supply is not sufficient to meet the current fuel wood demand (Banks *et al* 1996), although this might change as more households slowly switch to cooking with electric stoves (Madubansi and Shackleton 2007).

2. Materials and methods

2.1. Study area

The study area is located in the Lowveld of the savanna biome in the north-eastern part of South Africa (figure 1). The Lowveld is a low-lying landscape extending between the footslopes of the Drakensberg Great Escarpment to the west and the Mozambique coastal plain to the east (Venter *et al* 2003). The mean annual precipitation is approximately 630 mm yr^{-1}, with 25% inter-annual variability. Mean annual temperature is 22 °C. Dominant geology includes granite and gneiss with local intrusions of gabbro (Venter *et al* 2003). Vegetation communities are classified as 'granite lowveld' or 'gabbro grassy bushveld', according to the dominant underlying geology and the unique characteristics of the associated soils (Mucina and Rutherford 2006).

The three dominant land tenure systems are (i) state-owned conservation (Kruger National Park, KNP), (ii) privately owned conservation (Sabi Sand Game Reserve, SSGR) and (iii) state-owned communal areas (figure 1). The Kruger National Park was officially proclaimed in 1926 and today is the largest conservation area in South Africa covering 2 million hectares. Sabi Sand Game Reserve (SSGR) is a 63 000 ha privately protected area sharing a boundary with KNP on its southern and eastern limits. The reserve was officially established in 1965, prior to which the land was primarily used for commercial cattle farms. Today SSGR is an association of freehold owners with a strong tourism-based approach to conservation. The state-owned communal lands of the Bushbuckridge municipality are part of the former self-governing territories or 'homelands'. People largely rely on a combination of farming, livestock husbandry, and consumption and/or trade on informal markets of various natural resources (Shackleton *et al* 2001). Subsistence rain-fed crops are grown at the homestead or in arable fields located in the proximity of the settlements.

2.2. LiDAR-based biomass maps

Discrete-return LiDAR data were acquired in April–May 2008 with the Carnegie Airborne Observatory (CAO) Alpha system (Asner *et al* 2007), across the study sites covering 30 000 ha. Maps of aboveground biomass were created from the vegetation height maps using field calibration methods (for details see Colgan *et al* (2012)). In brief, biomass was first estimated in 206 field plots (30 m diameter, 0.07 ha) from basal stem diameter of all woody stems using a generalized allometric equation for South African trees (Nickless *et al* 2011). These plot-level biomass estimates were then correlated with the LiDAR metric $H \times$ CC, where H is the mean top-of-canopy height of each plot and CC is the plot canopy cover (Colgan *et al* 2012). The general $H \times$ CC model accounted for 75% of biomass variance ($R^2 = 0.75$) with a residual error of 48% of the mean biomass ($n =$ 101 plots). To account for differences in vegetation due to underlying substrates and soils, a second calibration equation was used for gabbro landscapes ($n = 27$, $R^2 = 0.82$, residual error = 48%). The final output resolution of the biomass maps (27 m × 27 m pixels) was selected to match the equivalent field plot area (0.07 ha). Figure 2 is an example of the final biomass map output, overlaid onto CAO hyperspectral imagery as reference.

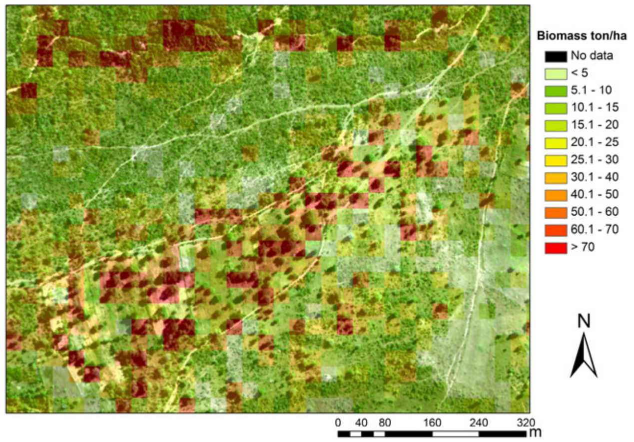

Figure 2. LiDAR-derived biomass estimates per 27 m pixel displayed transparently over a true colour image from the CAO imaging spectrometer (1.1 m resolution).

2.3. Supply–demand modelling

The study area forms part of the Agincourt Health and Demographic Surveillance System (AHDSS) managed by the Medical Research Council (MRC)—Wits Rural Public Health and Health Transitions Research Unit. An annual census of 21 villages (increased to 24 in 2007), covering a population of 14 000 households and 84 000 people has been conducted since 1992 (Kahn *et al* 2007), thus providing accurate longitudinal demographic data. The average household uses 3–4 tonnes of fuelwood per year (Madubansi and Shackleton 2007, Banks *et al* 1996, Shackleton *et al* 1994), which amounts to 156 630–208 840 t in the Bushbuckridge municipality (figure 1, inset).

Supply–demand calculations were performed for a selected communal landscape (figure 1, L5) where the supply and demand could be closely linked. The communal nature of rangelands in general makes it difficult to assign observed resource extraction to a particular village. However, for the selected village, Justicia, the entire demand for fuelwood could be applied to this site, due the fact that the rangeland is fenced on two sides by SSGR and is too far from other villages to allow collection on foot. These fences, together with the position of the village at the entrance to the rangeland, create an enclave where informal access control is possible. Previous household surveys confirmed that people from Justicia largely have exclusive access to this site, which is maintained by the local traditional authorities (Tuinder 2009). The supply–demand model could not be applied to the entire study area as wall-to-wall LiDAR data were not available, and most other villages do not have the option to control access to their rangelands to the same extent as Justicia.

The model presents a 'best-case-scenario' by assuming high values in the documented range of supply-side variables. Fuelwood demand was determined from a survey of 94 households in Justicia in 2010. 67% of households in Justicia still use fuelwood exclusively, while 33% sometimes use electricity as supplement. Given that Justicia currently has 1279 households, of which 67% consume approximately 3.5 t per household per annum, this village extracts a minimum of 3000 t of fuelwood from the landscape in year one. Various future scenarios of reduction in fuelwood consumption are simulated (see section 2.4). The number of households in Justicia was assumed to increase at an average 1.8% per year, based on 1992–2010 surveys. We furthermore assumed an average annual woodland productivity of 4% of biomass (Rutherford 1978, Banks *et al* 1996). Coppice regrowth makes a significant contribution to fuelwood and is actively managed for this purpose in communal areas (Twine 2005, Neke *et al* 2006). Banks *et al* (1996) previously underestimated annual recruitment and coppice regrowth at 20 kg ha^{-1} yr^{-1}. Based on recent field experiments in the area, for one of the most abundant coppicing species, *Terminalia sericea*, we calculated coppice regrowth at 89 kg ha^{-1} yr^{-1} (Twine 2012). The area of coppiced vegetation was estimated from the LiDAR vegetation height data between 1.5 and 2.5 m, which amounted to 3% of the L5 landscape.

Basic settlement-specific, supply–demand calculations were performed based on the LiDAR-derived standing biomass and survey data (Banks *et al* 1996). Rate of change in standing crop of biomass for an area (S) was expressed as:

$$\Delta S/\Delta T = f(S) + g(A) - h(w, P, z) \tag{1}$$

where,

Supply: S = standing crop biomass in area = 28023 t; $f(\,)$ = function: woodland productivity (tonne ha^{-1} yr^{-1}); $g(\,)$ = function: rate of woodland recruitment (tonne ha^{-1} yr^{-1}); A = site area = 1568 ha; r_e = recruitment and coppice growth = 0.089 ha^{-1} yr^{-1}.

Demand: P = number of households = 1279; k_p = annual increase in number of households = 1.8%; z = annual mean fuelwood consumption per household = 3.5 t $household^{-1}$ yr^{-1}; w = percentage of households using fuelwood exclusively = 67% (current); $h(\,)$ function: the rate of fuelwood removal (kg yr^{-1}).

Growth rate in terms of annual percentage biomass increase (k_b):

$$k_b = 4\% \qquad f(S) = \ln(1 + k_b)S. \tag{2}$$

The contribution to increase of standing crop from recruitment and coppice regrowth is:

$$g(A) = r_e A = 4.19 \text{ t yr}^{-1}. \tag{3}$$

2.4. Future scenarios

Annual standing biomass in the landscape was calculated for different scenarios, where the percentage of households using fuelwood exclusively (w, currently 67%) was reduced at varying rates (v). During the past ten years, data from the study area suggest that households have switched to cooking with electricity at a rate of 2% per year ($v = 0.02$) (Twine 2012). The impact of potential future interventions to significantly reduce fuelwood consumption to sustainable level was also modelled. These included, an annual reduction of 10% or 15% in households using fuelwood, initiated after a fixed period of two years, for a number of years (T), until a target of, for example, 22% of households using fuelwood (w_{target}) is reached, with the other 78% switching to electricity. Equation (4) was expanded to:

$$\Delta S_n = \Delta T f(S_{n-1}) + \Delta T g(A) - \Delta T h(w_{n-1}, P, z). \tag{4}$$

Time (ΔT) was discretized into annual time steps n such that,

$$\Delta T = 1.$$

Thus,

$$S_n = \Delta S_n + S_{n-1}. \tag{5}$$

Therefore,

$$n^* = \frac{\ln\left(\frac{w_{target}}{w_{current}}\right)}{\ln(1+v)} \tag{6}$$

n^* = number of years to reach w_{target}, rounded to the next largest integer. Equation (5) is applied successively for years $1 - n^*$

$$\begin{aligned} w_{current} &= 67\% \text{ of households} \\ w_{target} &= 22\% \text{ or } 33\% \text{ of households} \\ v &= \text{annual rate of reduction in fuelwood use} \\ \text{currently } v &= 0.02, \end{aligned}$$

annual rate of reduction in fuelwood use due to simulated intervention: $v = 0.1, 0.15$.

3. Results and discussion

3.1. Biomass in communal and neighbouring conservation landscapes

Some of the savanna landscapes (figure 1) differed notably in biomass due to their underlying geology and land-use history. On granite substrates, the communal landscape (figure 3(a), L7) contained an average of 12 t ha^{-1}, which was less than half the biomass of the conservation sites (figure 3(a), L1, L2, L3). Comparing these results to studies looking at fire and elephant impacts on savanna biomass stocks (Levick *et al* 2009, Smit *et al* 2010, Asner *et al* 2009), our results suggest that human-driven fuelwood extraction has a greater impact on biomass than do elephants and fire (Wessels *et al* 2011). The communal fields on granite substrates still contained an average of 10–13 t ha^{-1} biomass (figure 3(a), L6, L7), which

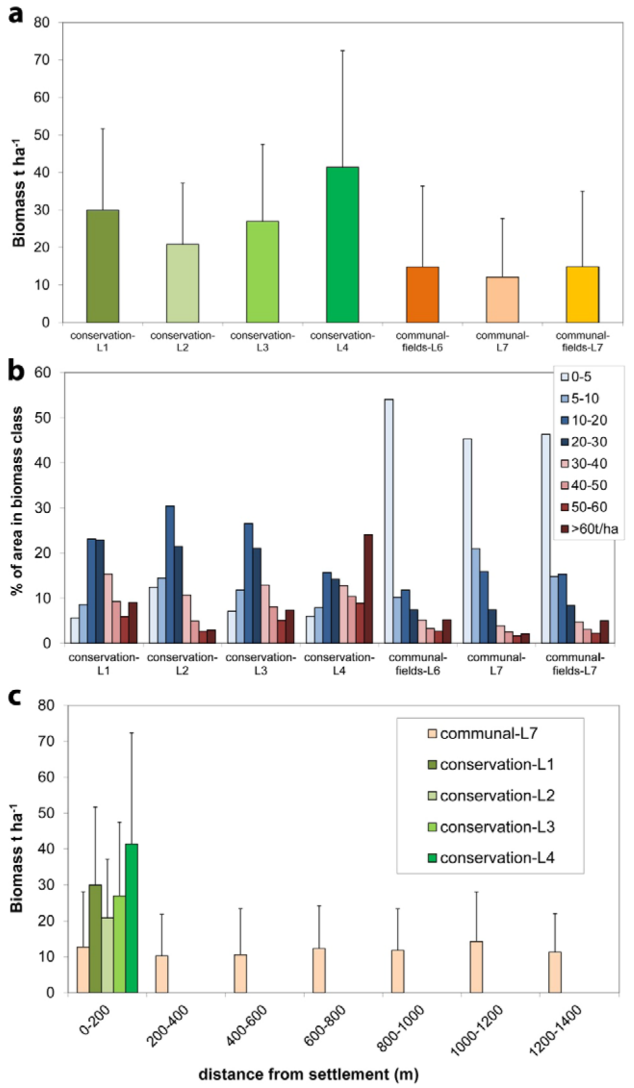

Figure 3. Biomass patterns of landscapes on granite substrates. (a) Mean biomass in all 27 m pixels inside each site. Error bars indicate standard deviation. (b) Percentage of each site containing specified amounts of aboveground biomass. (c) Mean biomass inside incremental 200 m buffer zones from settlements. Conservation sites are included as reference in 0–200 m class. Error bars indicate standard deviation.

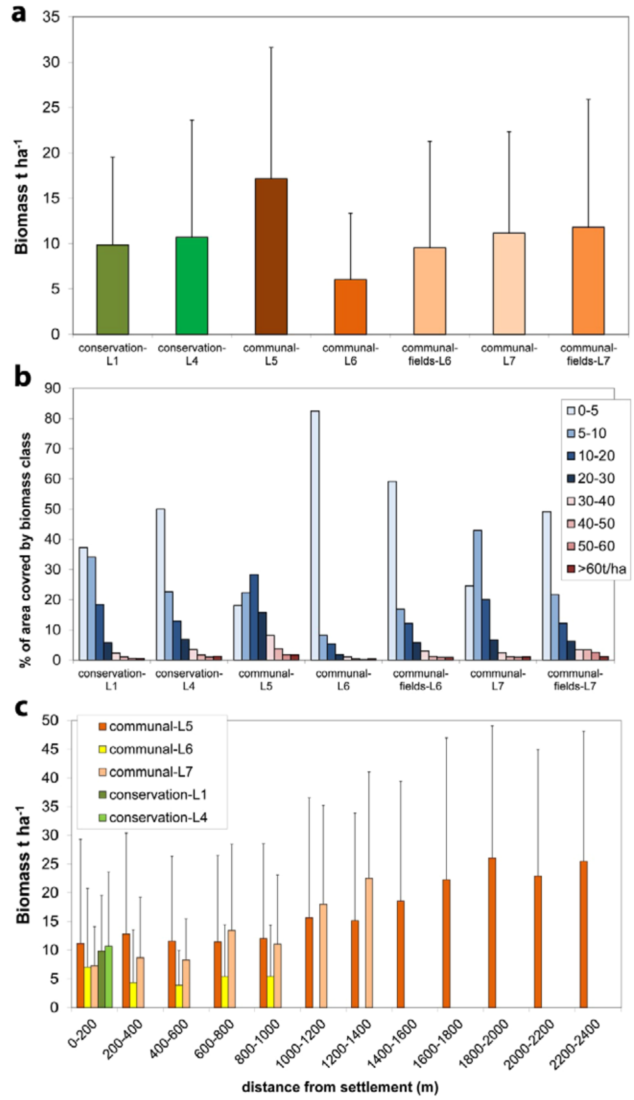

Figure 4. Biomass patterns of sites on gabbro substrates. (a) Mean biomass in all 27 m pixels inside each site. Error bars indicate standard deviation. (b) Percentage of each site containing specified amounts of aboveground biomass. (c) Mean biomass inside incremental 200 m buffer zones from settlements. Conservation sites are included as reference in the 0–200 m class. Error bars indicate standard deviation.

is mainly due to large fruiting trees protected by villagers for multiple non-timber uses (Shackleton *et al* 2003) (figure 2). The communal landscapes, including fields, had less than 5 t ha^{-1} across half of their area (figure 3(b), L6, L7). The adjacent conservation area had the highest biomass, located in a riparian floodplain (40 t ha^{-1}) (figure 3(a), L4).

On the gabbro substrate two adjacent communal savannas had highly contrasting average biomass (figure 4(a), L5, L6). More than 80% of one site had less than 5 t ha^{-1} biomass (figure 4(b), L6), whereas the majority of the other site (figure 4(b), L5) had between 10 and 20 t ha^{-1}. This could be attributed to the high utilization of the first site due to its proximity to two villages (<1000 m), while the latter is accessible by only one distant village (Justicia) as it shares a border with the private game reserve (figure 1).

Within communal savannas on gabbro substrates, areas that were more than 1 200 m away from settlements had almost double the biomass of the conservation sites (figure 4(c)). A fence-line effect was clearly visible along the boundary between SSGR and the communal area to the west, where the biomass inside the reserve is 6.5 t ha^{-1} (±9.6) and 17.1 t ha^{-1} (±16.3) in the communal savannas (figure 5). The high biomass in this communal landscape is likely due to the limited accessibility for fuelwood extraction, absence of elephants and differences in fire management (Levick *et al* 2009, Asner *et al* 2009, Eckhardt *et al* 2000, Van Wilgen *et al* 2004). The SSGR has been impacted by a 16-fold increase in the elephant population since 1993, when fences between SSGR and KNP were removed, leading to elephant-induced mortality of large trees (Shannon *et al* 2008a, Helm *et al* 2009, Eckhardt *et al* 2000).

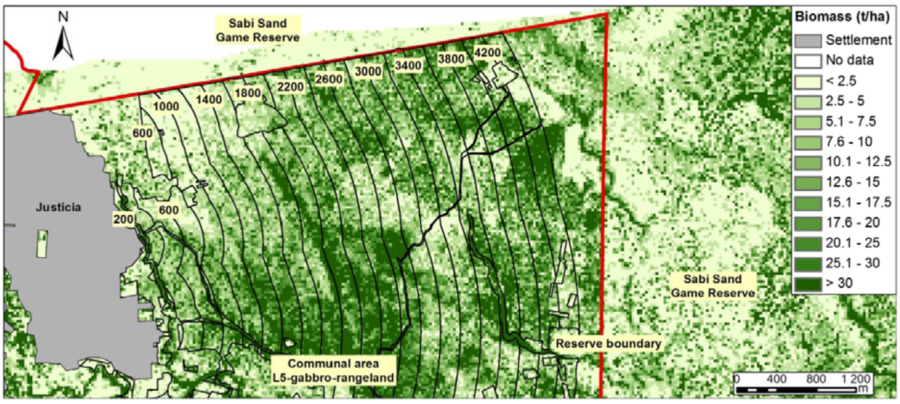

Figure 5. Aboveground biomass (tonnes per hectare–t ha^{-1}) in communal savannas east of the village of Justicia, overlaid with 200 m buffer zones. A clear fence-line effect on biomass is evident with higher biomass in the communal landscape (left) than in the Sabi Sand Game Reserve (right).

Interestingly, the high biomass (17–20 t ha^{-1}) in communal savannas far from settlements approached that of the long-term mammal exclosures inside KNP on similar substrates. For example, at a well-known experimental savanna site (Levick *et al* 2009), where large mammals have been excluded for 34 years and fire suppressed for seven years, LiDAR-measured biomass inside the exclosure was 36.9 ± 7.1 t ha^{-1} compared to 2.7 ± 0.5 t ha^{-1} outside (Colgan *et al* 2012). This suggests that where fuelwood extraction is low due to limited accessibility, biomass reserves on gabbro substrates are roughly equal to those found in conversation areas protected from large herbivores and reduced fire frequency (Smit *et al* 2010, Asner *et al* 2009). Therefore, the woodlands clearly have the potential to produce high biomass.

3.2. Disturbance gradient in biomass

Biomass increased with distance from settlements (figure 4(c), L5, L7), confirming the existence of disturbance gradients previously described in the study area based on limited field work (Shackleton *et al* 1994, Banks *et al* 1996). The one communal landscape consistently had approximately 10 t ha^{-1} up to 1000 m from settlements, after which the biomass steadily increased to 25 t ha^{-1} at 2400 m (figure 4(c), L5). Another communal landscape followed a similar trend—averaging 8 t ha^{-1} between 0 and 600 m, after which biomass increased to 22 t ha^{-1} at 1400 m (figure 4(c), L7). In general, the biomass increased at 2 t ha^{-1} for every 100 m beyond 1000 m from the settlements. (Note that this assessment excluded cultivated fields near settlements.)

It was previously found that biomass near settlements was only 13–17% of that measured 1000–2600 m away (Shackleton *et al* 1994, Banks *et al* 1996). This correlates with documented increases in walking distance to collect wood from 100 m in the 1980s to approximately 1000 m in the 1990s (Giannecchini *et al* 2007). The current results suggest that 'self-collection' of fuelwood is driving the pattern, since previous research has shown that collectors are not prepared to walk much further than 1000 m to gather fuelwood (Giannecchini *et al* 2007, Twine 2005), which explains the marked increase in biomass observed beyond 1000 m (figure 4(c), L5, L7). As a result, the percentage of households collecting wood themselves has decreased from 79% in 1991 to 68% in 2002, as households have switched to purchasing wood from vendors who collect wood using vehicles (Giannecchini *et al* 2007, Twine 2005, Madubansi and Shackleton 2007). Similar phenomena occurred throughout Africa as the harvesting frontier has moved beyond the reach of self-collectors (Aron *et al* 1991, Hiemstra-van der Horst and Hovorka 2009).

There were no disturbance gradients in highly impacted gabbro- (figure 4(c), L6) and granitic-savanna landscapes (figure 3(c), L7). This was due to their overall close proximity to multiple settlements and high extraction impacts up to 1200 m from settlements, which has reduced the biomass to less than half that of conservation sites. Due to heavy utilization by several surrounding villages, the disturbance gradients may have coalesced and subsequently disappeared over time (Fisher *et al* 2012). The current gradients may, however, disappear in the future, should harvesting using vehicles intensify (Fisher *et al* 2012, Twine *et al* 2003).

3.3. Supply–demand modelling

The annual future standing biomass in one of the communal landscapes (L5) was calculated using the models and parameters from Banks *et al* (1996). The village of Justicia was selected for the model application since its residents collect wood exclusively from the adjacent landscape (L5) and no other villages have access to this area (figure 1, L5) (Tuinder 2009). The fuelwood demand of this village could thus be entirely assigned to a specific area supplying the biomass. Figure 6 illustrates the potential reduction in biomass resulting from different future scenarios of fuelwood consumption. Under the current rate of fuelwood consumption (67% of households) and an observed 2% reduction in the number of households using fuelwood, all biomass in this communal landscape would be depleted within thirteen years (figure 6). However, should the quality (e.g. stem diameter)

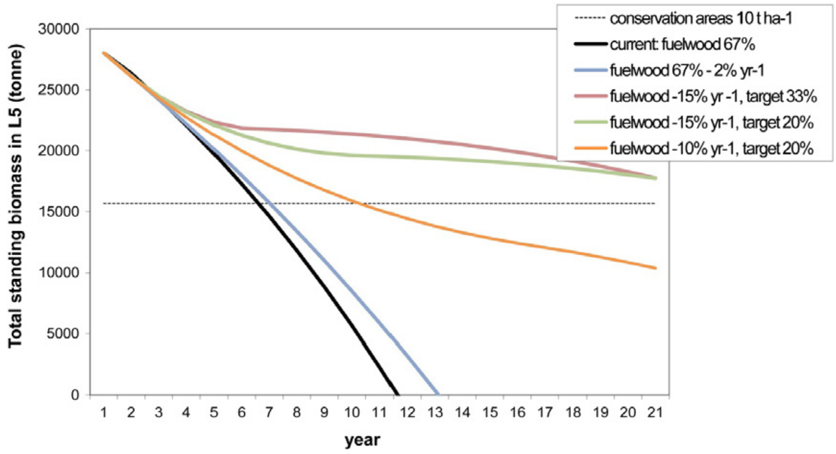

Figure 6. Total aboveground biomass in communal landscape predicted by supply–demand model for successive years, under different scenarios regarding reductions in the percentage of households using fuelwood exclusively. Fuelwood 67%–2% yr^{-1}: current situation where 67% of households use fuelwood with an annual reduction of 2%. Fuelwood −15% yr^{-1}, target 33%: future scenario where an intervention reduces the number of households using fuelwood by 15% per year until a target of 33% of households is reached. Annual reductions of 15% and 10% in combination with a target of 20% of household are also given. All future scenarios assume that the reductions in fuelwood use starts after two years. As a reference the total biomass expected in the area at the densities observed in conservation areas (10 t ha^{-1}) is also indicated.

and density of fuelwood become too low to make harvesting worthwhile, fuelwood collection may even cease 2–3 years before all biomass is depleted (Shackleton 1993). Recently Matsika *et al* (2012) compared their latest field data with that of Banks *et al* (1996) for two settlements just north of the current study area and found that biomass had reduced by 40% and 12% around the two respective villages within 17 years. The woodlands with the 40% reduction have become degraded and no longer produce fuelwood of preferred species and stem size in sufficient quantity or quality. Residents have thus been extracting fuelwood from an unoccupied neighbouring private farm for several years (Matsika *et al* 2012). Residents of Justicia in the current study may try to adapt their use patterns, but since the woodland under consideration has the largest total biomass reserve available to any village in the study area, there will be little opportunity to collect fuelwood elsewhere.

Unless more households start using electricity, the current demand will cause a significant reduction in biomass in the foreseeable future. Therefore, the impact of potential future intervention aimed at reducing fuelwood consumption to sustainable biomass levels was also investigated. The first mitigation strategy assumed a 15% annual reduction in the number of households using fuelwood for five years and then remaining constant at a target of 33% of households (compared to the current 67%) thereafter. This strategy will still result in a rapid reduction in biomass below 6275 t from 28 023 t (4 t ha^{-1}) within 20 years (figure 6). However, if the same intervention of 15% reduction was applied for an additional three years until only 20% of households use fuelwood, the biomass will start stabilizing at 18 000 t (11.5 t ha^{-1}) within 20 years, although a slight downward trend continues due mainly to the human population growth. Although not presented here, the model predictions are very sensitive to delays in the start of reductions in fuelwood consumptions (Banks *et al* 1996). Here, all future scenarios assumed that the reductions in fuelwood use starts after two years.

Our estimated rates of biomass depletion are conservative, because (i) the supply could be overestimated by the inclusion of large fruiting trees that are not harvested for fuelwood (Shackleton and Shackleton 2004, Shackleton *et al* 2002a, 2003), (ii) the transformation of woodlands to agricultural fields and settlements were not accounted for (Giannecchini *et al* 2007, Coetzer *et al* 2010), and (iii) there is new evidence of additional harvesting pressure from non-local harvesters (Twine *et al* 2003). In the future, changes in biomass resulting from all of these factors can be monitored with LiDAR data acquired every few years. Methods are currently being finalized to identify large fruiting trees using hyperspectral CAO imagery (Cho *et al* 2010) so that these trees may be excluded from biomass calculation to provide a more realistic estimate of fuelwood availability.

Fuelwood harvesting becomes unsustainable if it causes persistent changes in the woodland biomass such that the quality of fuelwood is diminished for a length of time that negatively impacts human well-being, resulting in a decline in social and economic capital (Shackleton *et al* 1994, Scholes 2009). The fuelwood crisis predicted for developing nations in the 1970s (Eckholm 1975, de Montalambert and Clement 1983) has not yet materialized, primarily due to a mismatch between the scale at which shortages occur and the scale at which global fuelwood accounting was done at the time (Dewees 1989). Fuelwood shortages are now regarded as localized phenomena and the novel combination of LiDAR-based biomass estimates and surveys of household consumption presented here, provide realistic village-specific, supply–demand calculations at the appropriate scale. The predictions of the supply–demand model echo concerns that these areas may be over-utilized to the point at which the ecosystem would be unable to sustain the services upon which the local people's subsistence livelihoods depend (Higgins *et al* 1999, Shackleton and Shackleton 2000, Banks *et al* 1996). This is exacerbated by important socio-economic trends, such as the weakening of traditional institutions which are responsible for woodland resource governance, increasing commercialization of the wood trade, and persistent poverty which hinders household ascension up the 'energy ladder' (Twine 2005).

4. Conclusions

The switch to electricity is still primarily limited by the costs of purchasing a stove and paying for the electricity, which remains unaffordable to most poor rural households (Madubansi and Shackleton 2006, Shackleton *et al* 2002b). Therefore, policies and interventions that promote the diversification of affordable energy alternatives, local-level management of woodland resources and rural economic development are needed. Failing this, the results of the present study suggest that the long-anticipated fuelwood crises will become a reality with dire ecological and socio-economic consequences (Dovie *et al* 2002, von Maltitz and Scholes 1995, Williams and Shackleton 2002, Banks *et al* 1996, Higgins *et al* 1999). On the other hand, the current situation may present an opportunity for rural communities in developing African countries to 'leapfrog' past fossil fuel dependency to more efficient and renewable energy sources (Hiemstra-van der Horst and Hovorka 2009).

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Appendix. Quantifying the value of the ecosystem service of fuelwood provision by savanna

The value of the ecosystem service of fuelwood provision by the savanna woodlands can be calculated from the national savings in electricity (MW) and expenditure on electricity generation (US$). In order to calculate these savings, we follow (Howells *et al* 2006) in assuming that if people made the energy switch, the average household would cook for 45 min a day on a two-plate cooker with a maximum power rating of 1.5 kW, running at 50% of the rated full power. Further, we assume that 70% of the cooking would occur at peak periods ('peak coincidence factor') (Howells *et al* 2006). Based on these assumptions, the additional peak period demand by these 712 800 households, if they all switched to using electricity for all their cooking needs, would be 375 MW. This energy savings is 62% of the total generation capacity (600 MW) of a four-unit open gas turbine power station with a construction cost of USD230 million in 2007. Using figures for the total cost of new power generation per kWh for coal-fired and open cycle gas turbine power stations (USD0.1/kWh and USD0.5/kWh respectively) (Anonymous 2011), this translates into an annual cost saving of USD14.6–77.4 million per year for coal-fired and open cycle gas turbine power stations respectively.

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