What lies beneath: detecting sub-canopy changes in savanna woodlands using a three-dimensional classification method

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Keywords
Change detection; Ecosystem services; Fire; Geology; Land use; Local Indicators of Spatial Association; Monitoring; Savanna

Abbreviations
BBR = Bushbuckridge Municipality; CR = communal rangelands; GLP = gains loss and persistence; LCCS = land cover classification system; LiDAR = light detection and ranging; LISA = Local Indicators of Spatial Association; PA = protected area; SSW = Sabi Sand Wildtuin.

Nomenclature
Mucina & Rutherford (2006)

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Abstract

Question: Increasing population pressure, socio-economic development and associated natural resource use in savannas are resulting in large-scale land cover changes, which can be mapped using remote sensing. Is a three-dimensional (3D) woody vegetation structural classification applied to LiDAR (Light Detection and Ranging) data better than a 2D analysis to investigate change in fine-scale woody vegetation structure over 2 yrs in a protected area (PA) and a communal rangeland (CR)?

Location: Bushbuckridge Municipality and Sabi Sand Wildtuin, NE South Africa.

Methods: Airborne LiDAR data were collected over 3 300 ha in April 2008 and 2010. Individual tree canopies were identified using object-based image analysis and classified into four height classes: 1–3, 3–6, 6–10 and >10 m. Four structural metrics were calculated for 0.25-ha grid cells: canopy cover, number of canopy layers present, cohesion and number of height classes present. The relationship between top-of-canopy cover and sub-canopy cover was investigated using regression. Gains, losses and persistence (GLP) of cover at each height class and the four structural metrics were calculated. GLP of clusters of each structural metric (calculated using LISA – Local Indicators of Spatial Association – statistics) were used to assess the changes in clusters of each metric over time.

Results: Top-of-canopy cover was not a good predictor of sub-canopy cover. The number of canopy layers present and cohesion showed gains and losses with persistence in canopy cover over time, necessitating the use of a 3D classification to detect fine-scale changes, especially in structurally heterogeneous savannas. Trees >3 m in height showed recruitment and gains up to 2.2 times higher in the CR where they are likely to be protected for cultural reasons, but losses of up to 3.2-fold more in the PA, possibly due to treefall caused by elephant and/or fire.

Conclusion: Land use has affected sub-canopy structure in the adjacent sites, with the low intensity use CR showing higher structural diversity. A 3D classification approach was successful in detecting fine-scale, short-term changes between land uses, and can thus be used as a monitoring tool for savanna woody vegetation structure.

Introduction

The effects of biodiversity loss on ecosystem function and services have been a major focus of global change research (e.g., Naeem 2002; Hooper et al. 2005; Balvanera et al. 2006; Hector & Bagchi 2007). Landscape modification and habitat fragmentation are two of the key drivers of biodiversity loss (Sala et al. 2000; Fischer & Lindenmayer 2007), with unsustainable natural resource use further exacerbating the problem. This issue is particularly pressing in savannas, which are home to over nine million rural poor in South Africa (Twine et al. 2003). Both the strong dependence on natural resources and expansion of settlements into intact vegetation have altered vegetation structure in this biome (Freitag-Ronaldson & Foxcroft 2003; Coetzer et al. 2010). Savannas are also prone to bush encroachment arising from changes in over-grazing/browsing intensity, over-harvesting and an unsuitable fire regime, resulting in an increase in the density of woody vegetation (Scholes & Archer 1997; Archer et al. 2001). Savannas, with their discontinuous woody layer in a continuous grassy matrix, are structurally heterogeneous. The mosaic of woody patches and complex vertical structure inherent in savannas make their structure difficult to quantify, especially over large extents. Monitoring of vegetation structure is essential in this biome for early detection of change in order to mitigate the potential negative effects of a reduction in resilience as a result of structural change and biodiversity loss.

Changes in woody vegetation structure are detectable both between land uses and over time. Traditionally, fine-scale measurements of structure such as tree height, stem diameter and number of stems are field-based, while large-scale but coarse measurements of structure such as woody cover and spatial patterns are often estimated using remote sensing methods. Time and financial constraints usually limit field surveys to measuring structure at one point or a few points in time. Although field measurements of structural dynamics are possible, they may not adequately portray the landscape variability, and they are often collected by different researchers using different protocols [e.g., woody structure in a riparian area in 1996/7 by Garner & Witkowski (1997) and in 2005 by Beater et al. (2008); and woody structure in two villages by Banks et al. (1996) in 1992 and Matsika et al. (2013) in 2009]. Matsika et al. (2013) found a reduction in wood stocks in two villages over time, although the finding was more pronounced in one village where fuelwood harvesting was unsustainable and the rangeland was being encroached by the settlement. The differences in rate of decline indicate patterns are settlement-specific, highlighting the need for change detection studies to be carried out over more extensive areas.

Remote sensing is necessary for long-term change studies over large regions or in areas that have not had fieldwork previously done in them. Giannecchini et al. (2007) conducted a 23-yr historical analysis of woody cover change (percentage cover and number of woody patches) for three villages using aerial photographs. The results were site-specific and related to intensity of use, population density, natural resource availability, diversification of livelihood strategies and drought, the findings of which support Matsika et al. (2013). Aerial photographs and satellite imagery such as Landsat are commonly used for change detection studies (e.g., Asner et al. 2003; Luoga et al. 2005; Mwavu & Witkowski 2008; Brink & Eva 2009; Coetzer et al. 2010) as they date back to the 1930s (aerial imagery) and 1970s (Landsat), thus allowing long-term change to be measured (Buitenwerf et al. 2012). Passive remote sensing products can be used to detect more than just changes in canopy cover by including changes in life form, spatial distribution, leaf type and phenology and stratification such as in the land cover classification system (LCCS; Di Gregorio & Jansen 2000). However, plants below the canopy cannot be detected using these techniques. Therefore, if bush encroachment occurs, or if the distribution of vegetation size classes changes, the change would go undetected (Jansen & Di Gregorio 2002). Indeed, it has been acknowledged that in tracking the Aichi Biodiversity Targets (CBD: Convention on Biological Diversity), bush encroachment monitoring cannot be done in the absence of LiDAR (light detection and ranging) or large-scale field methods (Secades et al. 2014).

Airborne LiDAR provides a powerful middle scale between field data and satellite remote sensing. LiDAR is an active remote sensing technology that measures sub-canopy information at a fine resolution over large extents via measurement of laser travel time (Lefsky et al. 2002). As this technology is relatively new, historical change detection is not yet common; however, data collected now can be used as baseline information for future monitoring investigations. The Carnegie Airborne Observatory (CAO) Alpha System (Asner et al. 2007) collected LiDAR data across two land uses (communal rangelands and protected areas) in the Lowveld region of South Africa in 2008 and 2010. We use a three-dimensional (3D) woody structural classification (Fisher et al. 2014a) to investigate change in fine-scale woody vegetation structure over a 2-yr period across two different land uses (communal rangelands and a private protected area) to address the following: (1) what are the advantages of using a 3D over a 2D vegetation structural classification for detection of change over time, including sub-canopy structural change; and (2) how does human use of the landscape affect woody vegetation structural change?

Methods

Site description

The two study sites border one another on the boundary between Sabi Sand Wildtuin (SSW), a private game reserve, and the village of Justicia in Bushbuckridge Municipality (BBR) situated in Mpumalanga province, northeast South Africa (Fig. 1). The total area is 3300 ha (2034 ha in SSW and 1 266 ha in BBR). As the two sites border one another, they share essentially the same biophysical characteristics. Rainfall is predominantly in the form of convection thunderstorms, with mean annual precipitation of 650 mm, while the mean annual temperature is 21 °C, with hot summers and mild winters (Shackleton et al. 1994). Topography is undulating with an altitudinal range of 310–460 m a.s.l., and the geology in the region is granite with Timbavati gabbro intrusions. However, only gabbro was present in BBR while both gabbro and granite were present in SSW. Granite lowveld is the dominant vegetation unit in the area, with gabbro grassy bushveld and legogote sour bushveld also occurring (Mucina & Rutherford 2006). Typical woody plant species in the granite lowveld include: *Terminalia sericea*, *Combretum zeyheri* and *C. apiculatum* on the deep sandy toplands, and *Acacia nigrescens*, *Dichrostachys cinerea* and *Grewia bicolor* on the more clayey soils of the bottomlands. In the two other vegetation units, additional common species include *Sclerocarya birrea*, *Lannea schweinfurthii*, *Ziziphus mucronata*, *Dalbergia melanoxylon*, *Peltophorum africanum* and *Pterocarpus rotundifolius*.

Bushbuckridge consists of two former Apartheid homelands, Gazankulu and Lebowa (Thornton 2002), which were formed with the Native Land Act (No. 27) of 1913. Between 1972 and 2012 human population density increased in the area to 209 people km^{-2} (http://interactive.statssa.gov.za), with resulting increase in land utilization intensity and economic impoverishment (Pollard et al. 2003). In 1994 the region was divided into Tribal Trust Lands and governed by Tribal Authorities who determine zonation into residential, arable and communal areas for grazing and harvesting of timber and non-timber products. Subsistence livelihoods are practised, and land utilization tends to be more intensive near the villages (Shackleton et al. 1994; Fisher et al. 2012). Historically, cultural values of the people in the area meant harvesting of live trees [especially Marula (*Sclerocarya birrea*) trees] used for medicine, fruit and culturally important activities was discouraged; however, the demand for fuelwood and timber now overrides these values (Higgins et al. 1999) as they feel they have no alternatives in the face of high electricity prices and localized shortages of fuelwood (Kirkland et al. 2007). Characteristic of BBR's former homeland status, there is rampant unemployment (14% of adults

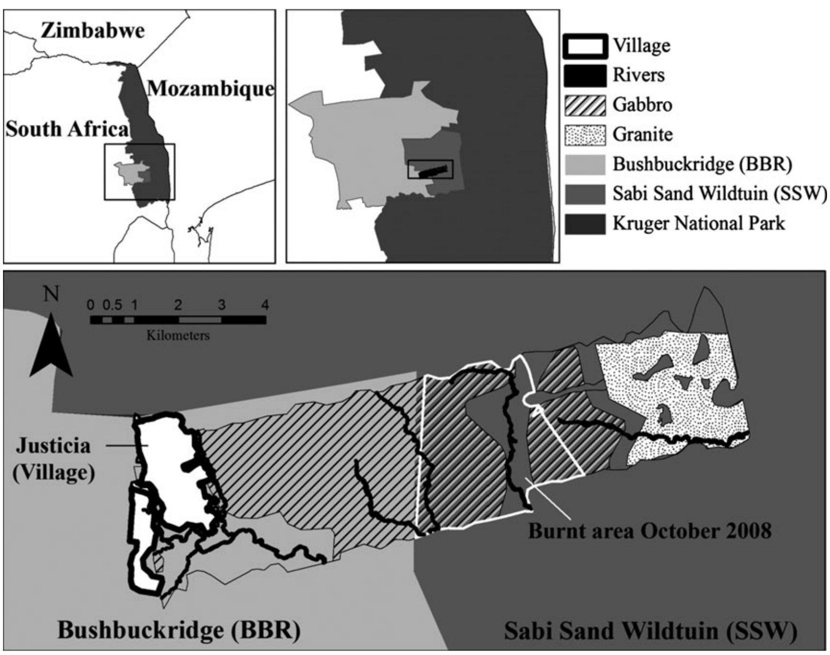


Fig. 1. Location of study sites within Bushbuckridge municipality (BBR) and Sabi Sand Wildtuin (SSW), Mpumalanga Province, South Africa. Justicia village, and granite and gabbro substrates are shown.

are employed; Phambili Energy 2009), poor infrastructure, high dependence on government-derived social grants and pensions, and reliance on migrant worker incomes (Shackleton et al. 2005; Madubansi & Shackleton 2007).

Sabi Sand Wildtuin is 65 000 ha, and was only formally proclaimed as a private game reserve in 1965. From 1922 to 1934, it was known as the Sabi Ranch, owned by the Transvaal Consolidated Lands (TCL), and was used for cattle ranching. Additional areas in the current SSW were purchased and used as game reserves around the same time, and in 1938 all cattle were removed due to a foot-and-mouth disease outbreak (J. Swart, pers. comm.). Each landowner within the conservancy manages their own land with regard to bush clearing and fire regimes. With the removal of fences between Kruger National Park and SSW in 1993, there was an influx of elephant into SSW, increasing from 0.0009 elephant ha^{-1} in 1993 to 0.007 ha^{-1} in 1998 (Hiscocks 1999). From 1996 to 1998, although the damage appeared high, only 21% of preferred tree species in southern SSW were damaged (Hiscocks 1999). Elephants primarily affect the structure rather than the species composition of trees, transforming vegetation to short woodland with a low density of large trees (Trollope et al. 1998). Structural changes are often better indicators of disturbance than compositional changes as, conversely, species richness can increase with disturbance (Shackleton et al. 1994). However, elephant do also tend to target certain species such as Marula, a keystone species, which has declined 25% in a 10-yr period (2001–2010) in Kruger National Park (Helm & Witkowski 2012). By 2010 the elephant population had increased to 0.013 ha^{-1}, although it has seen peaks of up to 0.02 elephant ha^{-1} in 2007 (the year prior to our first data collection) and, again, in 2012 (M. Grover, pers. comm.).

Light detection and ranging (LiDAR)

Woody vegetation was mapped across 3300 ha of semi-arid savanna in South Africa in April 2008 and April 2010 with the Carnegie Airborne Observatory Alpha System (CAO-Alpha; http://cao.ciw.edu). The CAO-Alpha combines imaging spectroscopy and LiDAR technologies for the study of ecosystems at a regional scale (Asner et al. 2007). The spectrometer was co-mounted with the LiDAR sensor, which collects both waveform- and discrete-return data; however, only the discrete-return data were used in this study. The data were collected at 2000 m above ground level with a laser pulse repetition frequency of 50 kHz, laser spot spacing of 1.12 m, and up to four returns per pulse (Asner et al. 2007). LiDAR produces a 3D xyz point cloud. The first canopy returns and last (ground) returns per 1.12 m grid cell were used to create a digital surface model (DSM) and digital elevation model (DEM), respectively. The DSM and DEM were created using triangulated models generated through linear interpolation of the LiDAR returns (van Aardt et al. 2006). The canopy height model (CHM) was then created by subtracting the DEM from the DSM. For 3D vegetation analysis, the frequency of returns of the point cloud was divided into volumetric pixels ($5 \times 5 \times 1$ m; X, Y, Z). The value in the voxel represents the frequency of LiDAR returns per 25 m^3 relative to the sum of returns for the entire 5×5 m vertical column (Asner et al. 2008) and was used to assess sub-canopy vegetation. Ground validation of the woody canopy heights (CHM) to compare to LiDAR-derived canopy heights was conducted concurrent to the airborne campaign in 2008 ($R^2 = 0.93$, $P < 0.01$, $n = 883$; Wessels et al. 2011).

The 3D classification of woody vegetation structure

A 3D characterization of woody vegetation is necessary to accurately measure structure, which in turn represents biomass, habitat and biodiversity as well as a metric of ecosystem services (Hall et al. 2011; Fisher et al. 2014a). Furthermore, a high degree of spatial detail is necessary to detect not only change but also modifications in land cover and vegetation structure. Jansen & Di Gregorio (2002) promote a parametric (classifier) approach to classification for change detection in line with the land cover classification system (LCCS). This type of approach allows for a consistent application of land-cover or land-use criteria, and a consistent use of criteria at the same level of classification, although actual criteria differ for each land-cover type, ensuring greater specificity and change detection ability (Jansen & Di Gregorio 2002). Fisher et al. (2014a) developed a 3D classification of savanna vegetation structure using principles taken from LCCS. The classification is specific to savanna vegetation and uses ecologically meaningful height classes related to fire, herbivory, frost and human use (Fisher et al. 2014a).

The first level of classification uses the CHM, and plants are delineated using object-based image analysis and classified as shrubs (1–3 m), low trees (3–6 m), high trees (6–10 m) or tall trees (>10 m; Fig. 2). Top-of-canopy cover is classified for 0.25-ha grid cells according to percentage canopy cover as measured on the CHM. Number of canopy layers present is calculated using the voxel data. Subsequently, cohesion of patches of the woody layer (using the canopy cover metric) and of different height classes (using the canopy layers derived from the voxel data) are calculated (Fisher et al. 2014a; Fig. 2). The second level of the classification categorizes the individual height classes within the canopy and sub-canopy using the voxel results from the LiDAR data analysis. Canopy layers are described in height order from shrub to tall tree. Here we explored changes in the metrics and the four height classes between land uses, geology and years.

Woody vegetation structure characterization

The advantages of a 3D over a 2D classification were investigated by comparing the gains (G; *increase* in the value of the metric under consideration), losses (L; *decrease* in the value of the metric) and persistence (P; *no change* in the value of the metric) (Coetzer et al. 2013) of the percentage of number canopy layers present (CL) and canopy cohesion with persistence in canopy cover from 2008 to 2010 (n = 13 198, 0.25-ha grid cells). Canopy cover was binned as follows: 0, 0.1–5%, 5.1–10%, 10.1–20%, 20.1–30%, 30.1–40%, 40.1–60%, 60.1–70%, >70% (Fig. 2; Fisher et al. 2014a); therefore gains, losses and persistence (GLP) were determined if there was a change in the cover class. For example, if the cover class changed from '5' (20–30%) to '6' (30–40%) a gain would be denoted. For continuous variables (not binned) such as cohesion (Fig. 2), the value of the metric had to exceed a change of >5% before it was considered a gain or loss of value in order to omit small changes that might be due to minor error (i.e., if a metric changed from 61.7% to 62.3% this would not be considered a gain). We investigated the relationships between the percentage cover of the four height classes as measured on the top-of-canopy image (i.e., seen from above), and the percentage cover of the sub-canopy vegetation within each height class (i.e., lateral view) in 2008 and 2010 using regression. Regressions were performed in R Studio (R v2.13.0; UsingR package; R Foundation for Statistical Computing, Vienna, AT). The effects of human use on woody vegetation structure over time were demonstrated by comparing change in BBR against changes in SSW.

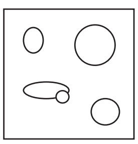
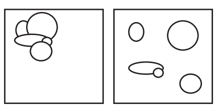
Functional metric	**Classification**	**Ecological relevance and description**
Canopy Cover (CC; categorical data) 	**Canopy cover** — **Cover (%)** Bare — 0 Grassland — 0.1-5 Sparse — 5.1-10 Open — 10.1-20 Open-Moderate — 20.1-30 Moderate — 30.1-40 Moderate-Closed — 40.1-60 Closed — 60.1-70 Forest — >70	Canopy cover is a key descriptor of biomes, with savannas having around 5-60% woody canopy cover (Mucina & Rutherford 2006). An increase or decrease in cover may be the result of a biome shift. The metric CC refers to the vertical projection of the tree/shrub crown onto the ground, given as a percent of the area. Cover is measured for the overall woody cover (all height classes). The dominant cover class is measured from the CC metric as the class that constitutes ≥50% of the total woody canopy cover. Canopy cover is measured from the top-of-canopy objects produced in eCognition based on the Canopy Height Model (CHM) LiDAR product.
Cohesion (continuous data) 	**COHESION** 0≤COHESION≤100 0 = no coverage 100 = continuous coverage	Measure of woody habitat connectivity (McGarigal et al. 2002). At a fine scale cohesion has implications for organisms' movement through, and use of, the landscape. At a landscape scale high cohesion would reduce edge effects (Fischer & Lindenmayer 2007). An increase in cohesion of one or more vertical height classes may indicate increased bushiness. The metric Cohesion is a measure of how aggregated the vegetation components (trees and shrubs) are within the designated area in the horizontal plane. Values range between 0 and 100, with 100 representing greater aggregation or clumping. Due to the mix of grass and woody components defining savannas, spatial arrangement is an important consideration with implications for habitat suitability and utilisation. Cohesion was measured for both the entire woody layer within the grassland matrix (using the CC metric), as well as for each height class as measured using the voxel data (i.e. canopy layers) to measure the cohesion of each height class.
Number of height classes (categorical data)	**Height classes** — **Range (m)** Shrub — 1-3 Low tree — 3.1-6 High tree — 6.1-10 Tall tree — >10	The greater the number of life forms present, the higher the structural heterogeneity. This may also increase faunal diversity as a result of increased habitat niches (Ishii et al. 2004). Higher diversity might also increase resilience to global change and/or intense use/management of the landscape (Fischer et al. 2006). The number of height classes present is calculated from the CHM layer.

Fig. 2. Ecological relevance of 3D woody vegetation structural classifiers and dynamics (after Fisher et al. 2014a).

Woody structural dynamics

Gains, losses and persistence of canopy cover, canopy cohesion and number of height classes present as measured from the voxel data (sub-canopy) and top-of-canopy layer (CHM) in SSW and BBR were compared. Spatial patterns of clusters of high and low values of canopy cover, canopy cohesion and number of height classes in 2008 and 2010 were quantified using a local indicator of spatial association (LISA) statistic, local Moran's I (Anselin 1995), in ArcMap 10.0 to identify ecologically meaningful clusters as opposed to small groups of similar values that may not have any ecological significance. Local Moran's I is used to assess the influence of locations on the magnitude of the global Moran's I statistic, with significance values giving a representation of the spatial clustering of similar values around each grid cell (Anselin 1995). The z-score (based on each metric's SD), P-value (probability of the observed pattern being created by a random process) and local mean value of the respective classification metric were calculated for each cell, and cells which were significantly different as determined using a permutation approach were classified as HH (High High), LL (Low Low), HL (High Low) or LH (Low High) as follows. In order to classify the grid cell, the target mean of that cell is compared to the local mean of neighbouring grid cells using an inverse distance spatial relationship (features that are closer together have a larger influence on the local mean than features further away). For grid cells with a strong positive z-score (>1.96), a cell is either classified as HH if the target mean is high and is surrounded by high values of the local mean, or LL if the target mean is low and surrounded by similar low values of the local mean. For spatial outliers (grid cells with z-scores < -1.96), grids cells are classified as HL if the target mean is higher than the local mean, and LH if it is lower (ESRI 2010). To simplify these classifications, they may be interpreted as follows: HH (highly significant clusters of high values), LL (highly significant cluster of low values), LH (outlier in which a low value is surrounded by predominantly high values) and HL (outlier in which a high value is surrounded predominantly by low values). Grid cells with no significant clusters or associations between outliers and clusters are classified as NS (not significant) (ESRI 2010). LISA statistics were calculated for BBR and SSW sites, as well as four subset areas, in 2008 and 2010: gabbro areas in BBR (BBR_Gabbro) and in SSW (SSW_Gabbro), granite areas in SSW (SSW_Granite) and the area in SSW that was burned in October 2008 (SSW_fire). To measure the change in the size and location of clusters, as defined using local Moran's I, of each metric over time, the gain, loss and persistence of clusters was calculated. Maps of Anselin's local Moran's I indicators of spatial association in 2008 (Fig. 3b) were subtracted from corresponding 2010 maps (Fig. 3a) to determine whether there was a gain (increase in spatial extent of significant clusters), loss (decrease in spatial extent of significant clusters) or persistence of any high or low clusters (no change in clusters/not significant cells did not become HH, LL, HL or LH; Fig. 3c).

Results

Woody vegetation structure characterisation

Grid cells with a persistent canopy cover (i.e., no change in total percentage canopy cover) showed gains, losses and persistence in the percentage number of canopy layers present and canopy cohesion from 2008 to 2010 (Fig. 4). Percentage number of canopy layers showed larger losses (2.3-fold more) and less persistence (three-fold less) compared to cohesion for areas with persistent canopy cover over the 2 yrs (Fig. 4).

The relationship between top-of-canopy cover (the cover of each height class as seen from above) and cover present within the canopy (as seen laterally) for the four height classes is not 1:1 (Fig. 5). Percentage cover of each height class is higher for the sub-canopy compared to the top-of-canopy for the respective height classes (Fig. 5), although differences are more pronounced from 1–3 and 6–10 m where the slopes of the regressions are ≤0.5 (Fig 5a,b,e,f). A significant relationship is present between top-of-canopy percentage cover and sub-canopy percentage cover from 3 to 6 m ($P < 0.0005$; $R^2 = 0.76$; Fig 5c; $R^2 = 0.82$; Fig 5d), which often constitutes the highest amount of cover in a grid cell, and > 10 m ($R^2 = 0.93$; Fig. 5g; $R^2 = 0.78$; Fig. 5h), often the lowest amount of cover. Higher sub-canopy cover than top-of-canopy cover is present from 1 to 3 m, indicating high density of vegetation within this height class, which is present under most other height classes. Even when no shrub layer is visible in the top-of-canopy (but other height classes are present and only tall vegetation is identified), >90% of the vegetation present may contain a shrub layer (Fig. 5a). The largest change in sub-canopy cover from 2008 to 2010 is in the 1–3 and >10 m height classes (Fig. 5a,b,g,h), with the sub-canopy cover showing more variation than top-of-canopy cover [reduction in R^2 from 0.4 to 0.35 (Fig. 5a,b) and 0.93 to 0.78 (Fig. 5g,h)].

Each height class showed higher persistence than gains or losses, although this result was more pronounced for height classes >6 m (Fig. 6c,d). Across all height classes and land uses, gains were systematically higher than losses. From 1 to 3 m, GLP were similar for SSW and BBR (Fig 6a), while SSW showed consistently higher percent-

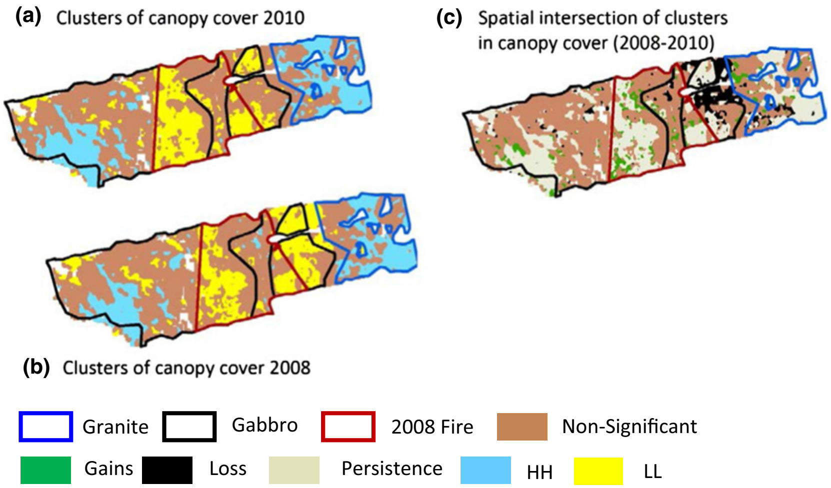


Fig. 3. Example of how gains, loss and persistence (GLP) of clusters of high/low values of a particular metric, in this case canopy cover, is derived. Anselin local Moran's *I* indicator of spatial association (HH: highly significant clusters of high values, LL: highly significant cluster of low values; NS: Not significant areas, i.e., no clusters) was calculated for canopy cover in (**a**) 2010 and (**b**) 2008. The differences between where clusters occur in 2008 and 2010 and are depicted in a GLP map (**c**) where gain indicates an increase in clustering, loss indicates a decrease in clustering and persistence is no change in clustering. NS indicates a persistence of no clusters.

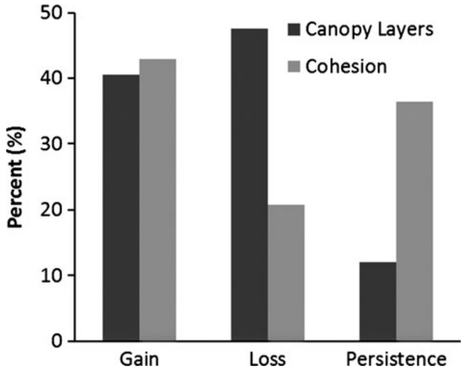


Fig. 4. Gains, losses and persistence (GLP) in the percentage canopy layers present and canopy cohesion in areas where canopy cover showed no changed (i.e., persistent canopy cover) from 2008 to 2010 in Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) study sites combined, South Africa (n = 6149, 0.25-ha grid cells).

age losses than BBR for height classes 3–6 m (2.75-fold higher), 6–10 m (3.2-fold higher) and >10 m (2.6-fold higher). Similarly, BBR showed higher percentage gains than SSW for height classes >6 m, particularly from 6 to 10 m (2.2-fold higher; Fig. 6b–d).

Structural dynamics across land use

Given the 2-yr time period, noteworthy gains and losses of value for each metric, and changes in how these metrics cluster, were observed (Figs 6–9). Although gains in canopy cover and cohesion occurred (Figs 7a, 8a, respectively), there were corresponding losses of significant clusters of canopy cover (Fig. 7b) and canopy cohesion (Fig. 8b) in SSW. The gains in cover and cohesion are a result of the gains in height classes <10 m (Fig. 6). There was a gain in significant clusters of all metrics (Figs 7–9) in the burned areas of SSW with corresponding losses in the value of the metrics (Figs 7–9) between 2008 and 2010. Gains and losses of statistically significant clusters predominantly occurred around existing clusters, i.e., existing clusters of a metric act as a nucleus of change.

Discussion

Structurally heterogeneous savannas necessitate a 3D approach to detect changes in vegetation structure, as a measurement of canopy cover alone would not indicate the changes that are occurring in the understorey (Fig. 5). The persistence in canopy cover, which would be regarded as no change over time using a 2D classification, is very different from persistence in both the vertical or horizontal domain of structure as measured by the number of canopy layers present and cohesion, respectively (Fig. 4). The vegetation cover seen in 2D, and the change in cover over the 2-yr time period, are not good predictors of sub-canopy cover (3D vegetation). Although phenology may affect results, the influences of this on the data were accounted for as far as possible by collecting the LiDAR in the same month during each campaign. Furthermore, the size of grid cell used and the 5% confidence interval of change when calculating gains and losses ensure our confidence in the results.

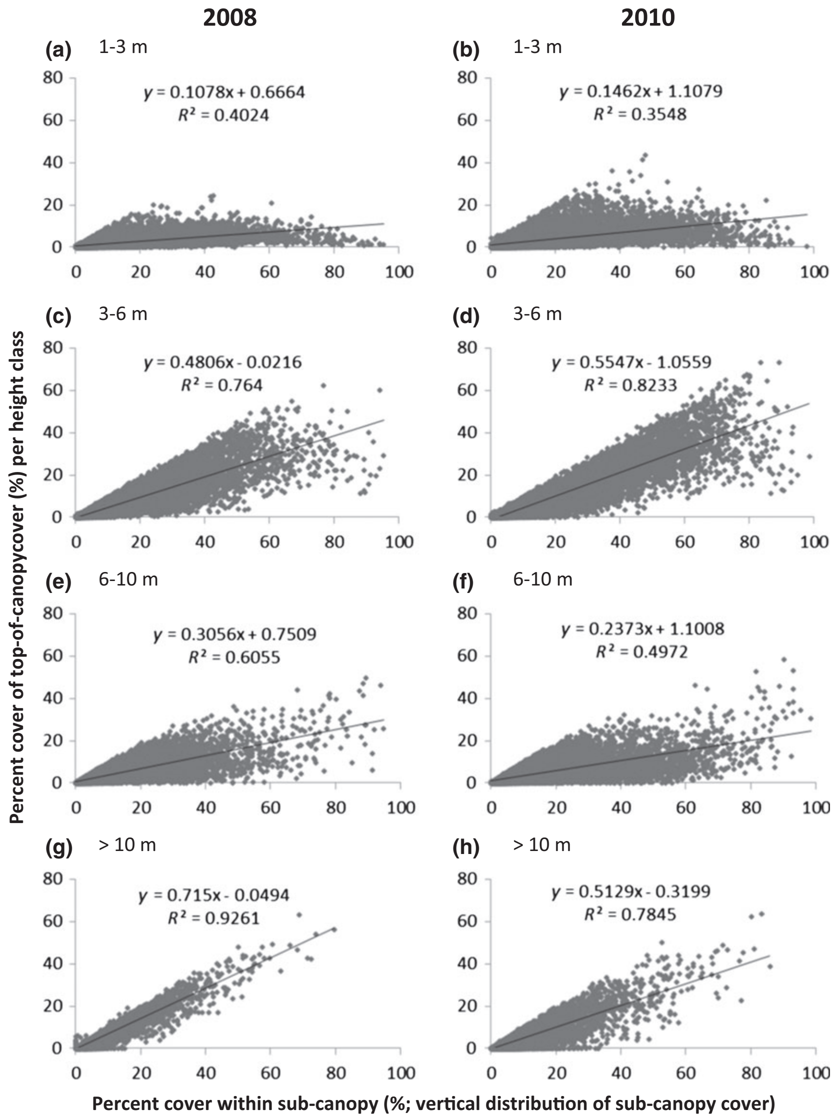


Fig. 5. Relationship between percentage total canopy cover from above for each height class and the percentage sub-canopy cover (lateral view) of vegetation present within each height class as measured using the voxel LiDAR data for four height classes (1–3, 3–6, 6–10 and >10 m) in 2008 and 2010 in Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) study sites ($P < 0.005$; $n = 13\ 198$ 0.25-ha grid cells).

The gains, losses and persistence of each height layer mirror the overall trend in top-of-canopy cover GLP (Fig. 6); however, when examining these patterns spatially, it becomes evident the majority of gains in SSW are below 6 m, whilst gains in BBR occur from 1 to 10 m. Furthermore, the gains in vegetation <10 m in BBR would be missed if only viewing changes in top-of-canopy cover (Fig. 6). These height-specific findings have implications for the management and use of the areas. Tall trees are protected in communal rangelands and special permission is needed to cut them down (Twine 2005). Coupled with recruitment into these taller height classes, the protection explains the gains in these height classes as well as the high percentage of persistence, especially in trees >10 m in BBR (Fig. 6d). Fuelwood and fencing poles are harvested from trees predominantly <3 m (Twine 2005; Neke et al. 2006), evident in the small amount of loss of vegetation from 1 to 3 m (Fig. 6a). The area of loss in the northeast of BBR in the 1–3 m class has been highlighted as a preferred location for fuelwood collection and poles as there is an abundance of thicker stems, particularly of *Acacia nigrescens* (Tuinder 2009). The gains observed in the percentage cover of shrubs are due to either coppicing or bush encroachment (Fig. 6a; Neke 2005). Savanna species have a strong regeneration response through coppice regrowth from harvested vegetation (Higgins et al. 1999; Kaschula

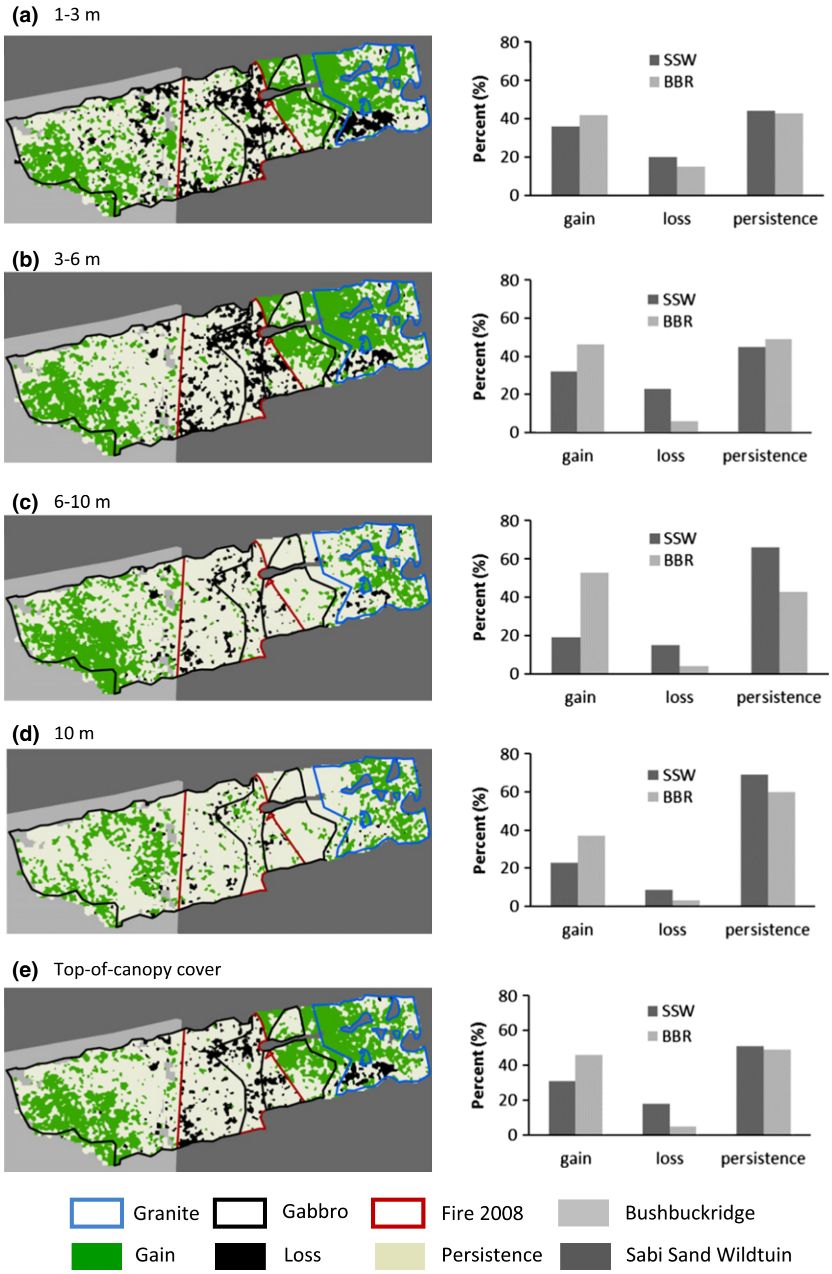


Fig. 6. Gains, losses and persistence (GLP) for four height classes (a. 1–3 m, b. 3–6 m, c. 6–10 m, d. >10 m) measured using volumetric pixels in 0.25-ha grid cells in Sabi Sand Wildtuin (SSW; n = 8136, 0.25-ha grid cells) and Bushbuckridge (BBR; n = 5062, 0.25-ha grid cells) study sites from 2008 to 2010.

et al. 2005). Bush encroachment is also prevalent in overgrazed savannas (Miller & Wigand 1994; Scholes & Archer 1997; Archer et al. 2001). Vegetation within the communal rangelands is therefore increasing with gains exceeding losses in all height classes. While the gains in vegetation do point towards densification of the woody layer, it also means there is a larger, and regenerating, wood supply for the rural community, provided the wood is of sufficient quality and quantity. Matsika et al. (2013) found the mean diameter of stems collected from rangelands in Bushbuckridge had significantly decreased between 1992 and 2009; indicating a reduction in the quality of stems available for fuelwood.

Similarly, height-specific maps indicate the percentage gain of vegetation from 1–3 and 3–6 m in SSW is almost equal to that of persistence (Fig. 6a,b), showing increasing woody vegetation density despite the effects of herbivory and fire, perhaps indicating bush encroachment. The area

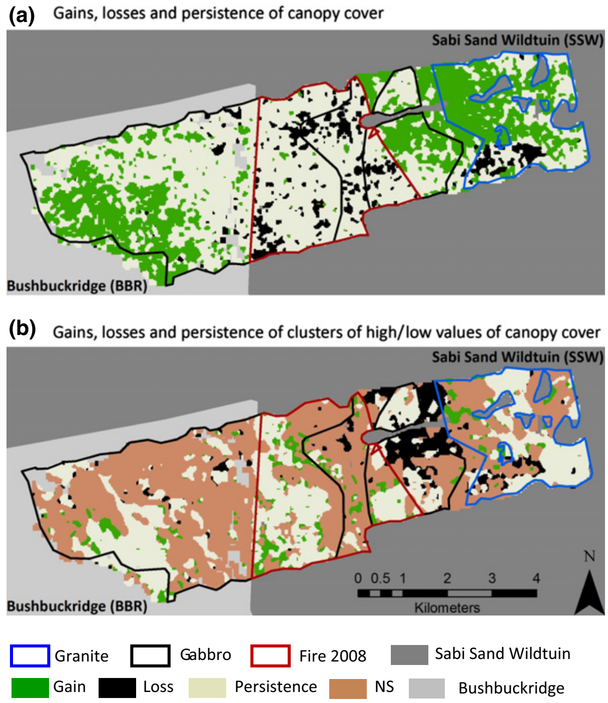


Fig. 7. Gains, losses and persistence of (**a**) canopy cover and (**b**) clusters of high/low values of canopy cover from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin local Moran's *I* indicator of spatial association. Gain indicates an increase in clustering, loss indicates a decrease in clustering and persistence is no change in clustering. NS indicates a persistence of no clusters.

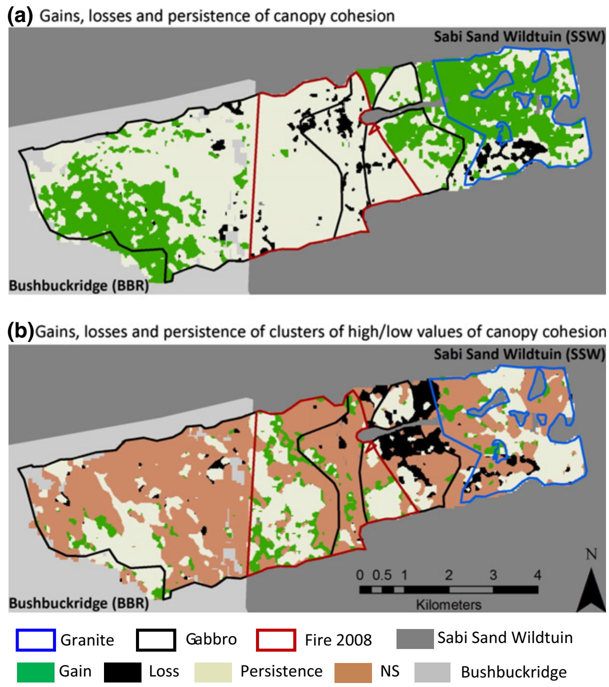


Fig. 8. Gains, losses and persistence of (**a**) canopy cohesion and (**b**) clusters of high/low values of canopy cohesion from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin local Moran's *I* indicator of spatial association. Gain indicates an increase in clustering, loss indicates a decrease in clustering and persistence is no change in clustering. NS indicates a persistence of no clusters.

is prone to bush encroachment as a result of previous cattle farming on the land (http://www.sabisand.co.za/ssw-history.html; Tobler et al. 2010; Papanastasis 2009). Fire is successfully used as a management tool in SSW as a result of the propensity towards bush densification. Canopy cover, cohesion and number of height classes showed losses within the burned area (Figs 7a, 8a, 9a). The decrease in canopy cover and canopy cohesion as a result of fire will affect how animals use the landscape, with most ungulates showing a preference for open spaces (Riginos & Grace 2008), which could increase predation on the congregated ungulates. This likelihood of increased predation is advantageous for SSW management, which receives revenue from tourism, namely game viewing, which is better in less dense bushveld.

Interestingly, the gains in vegetation in all height classes are higher in BBR than in SSW. The gains are predominantly on the western portion, adjacent to the settlement. Paradoxically, the gains in vegetation adjacent to the settlement were in the 1–3 and 3–6 m groups, whilst supply–demand models for the area have predicted a reduction in wood stocks (Banks et al. 1996; Wessels et al. 2013). Indeed, trees >3 m were up to 2.2-fold more in the communal rangelands than in the protected area, a possible indication of the protection for cultural reasons in BBR. Alternatively, this increase might be due to tall coppice growth with a dense canopy. Similarly, SSW showed losses of up to 3.2-fold more for trees >3 m as a result of treefall by elephant and fires (Fig. 6). SSW does show higher overall persistence in vegetation (Fig. 6), indicating more stable vegetation structure; however, this may point to reduced heterogeneity over time (Fisher et al. 2014b).

The gain and loss of the various structural metrics from 2008 to 2010 does not necessarily translate into gains or losses of clusters; rather, existing clusters act as nuclei around which new clusters will be formed or clusters will be lost. A gain in canopy cover, for example, might even mean a loss of clusters (see Fig. 7a,b – SSW) indicating a dynamic landscape, becoming more heterogeneous as clusters of similar vegetation cover are lost. A gain in clusters around existing clusters shows an aggregation of similar values (either high or low), indicating the loss of structural diversity within the landscape. Management interventions promoting heterogeneity should therefore focus around monitoring, and if necessary, eliminating

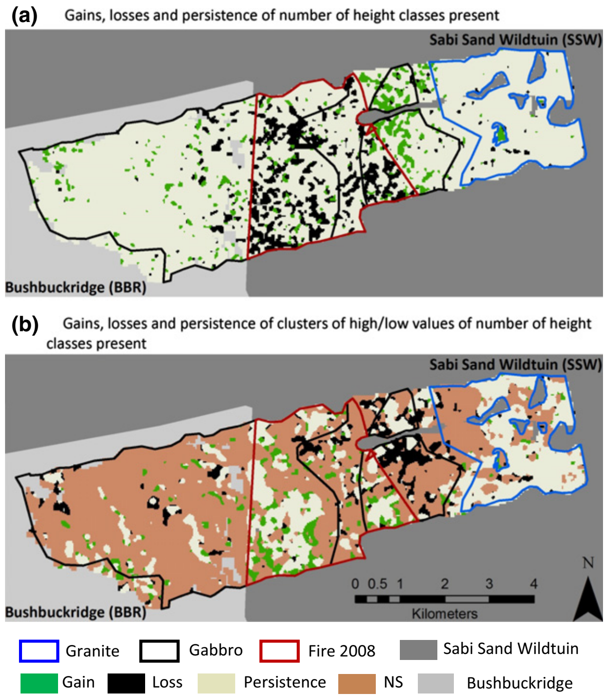


Fig. 9. Gains, losses and persistence of (**a**) number of height classes present and (**b**) clusters of high/low values of number of height classes present from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin local Moran's *I* indicator of spatial association. Gain indicates an increase in clustering, loss indicates a decrease in clustering and persistence is no change in clustering. NS indicates a persistence of no clusters.

clusters of similar vegetation, e.g., as occurs with bush encroachment. Patches of structurally similar vegetation are likely to be less resilient to both local and global change (van de Koppel & Rietkerk 2004).

Conclusions

A 3D classification approach ensures processes such as bush encroachment or sub-canopy structural changes are detected. These would otherwise be missed using a classification that only measures the aerial extent of cover (Jansen & Di Gregorio 2002). It is necessary to understand the structural implications of woody vegetation change, especially in the context of global bush encroachment trends (Kgope et al. 2010; Buitenwerf et al. 2012). Bush encroachment has many management implications related to cattle grazing, fuelwood provision, conservation of biodiversity and resilience. Despite government acknowledgement of the need for monitoring of woodland resources (National Forests Act No. 84 of 1998) and the threat of bush encroachment's effects on ecosystem service provision/biodiversity (Convention of Biodiversity), there is a lack of landscape-scale data, let alone 3D data required for bush encroachment monitoring. We clearly show that we can successfully monitor vegetation changes using a 3D classification applied to LiDAR data. Future work could be done to test these relationships across a greater variety of sites spanning a temperature and rainfall gradient.

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