Multi-temporal Remote Sensing Land-cover Change Detection for Biodiversity Assessment in the Berg River Catchment

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Abstract

Due to the intimate relationship that exists between land cover and biodiversity it is possible to draw inferences on the current state of the biodiversity of an area, assess the likely future pressures and plan accordingly based on an analysis of land-cover change. As a means of assessing the state of biodiversity in the Cape Floristic Region, two land-cover maps (1986/7 and 2007) were developed and demonstrated for the Berg River catchment in the Western Cape province of South Africa using multispectral Landsat Thematic Mapper (TM) data. The land-cover maps were produced to an accuracy of 85% using an object-orientated nearest neighbour supervised classification. The existing vegetation types of South Africa data set were superimposed on the newly classified remnants of natural vegetation to model changes in biodiversity. It was found that the area occupied by natural vegetation increased by more than 14%, suggesting an increase in biodiversity from 1986/7 to 2007. Considerable variation between vegetation types was, however, recorded. The land cover mapping, change analysis and biodiversity modelling methods employed by this study show that land-cover change analysis provides an ideal platform from which to initiate more intensive analyses of biodiversity change and conservation. Some limitations to the use of Landsat imagery for biodiversity monitoring are discussed.

1. Introduction

South Africa exhibits high rates of species diversity, richness and endemism due to the heterogeneity of topographic and climatic conditions, as well as its relatively large size. The country is home to three internationally recognized biodiversity hot spots, namely the Cape Floristic Region (CFR), the Succulent Karoo and Maputaland-Pondoland-Albany Thicket, as well as numerous threatened and endemic animal species (Thuiller et al., 2006). Over the last 100 years, the country has experienced significant land-cover changes as a result of human endeavour, which have had repercussions on the biodiversity (Biggs et al., 2002, Low et al., 1996, Mucina et al., 2006). Renosterveld with only 5% of its original extent remaining is one of the main habitat types in the CFR which contains extraordinary biodiversity being home to rare endemic flora (bulb species and other geophytes e.g. belonging to the *Iridaceae*, *Amaryllidaceae*, *Hyacinthaceae* plant families) (von Hase et al., 2003) as well as fauna (the critically endangered geometric tortoise *Psammobates geometricus* (Van Bloemenstein, 2005) and rare Cape dwarf chameleon *Bradypodion pumilum* (Tolley et al., 2010).

Anthropogenic land cover transformation threatens biodiversity directly through loss or degradation of habitat which causes species to be displaced or eradicated, impacting on species persistence and population viability (Haines-Young, 2009, Parker et al., 2002). The immediate effects of land-cover alterations on biodiversity are compounded by long-term hindrance of ecosystem functionality through degradation, fragmentation and contamination (Fahrig, 2001, 2003, Haines-Young, 2009, Parker et al., 2002). Since biodiversity and ecosystem functionality are intricately interrelated, disruptions to particular components of this relationship may have adverse effects on overall ecosystem integrity (Gaston, 1996, Wessels et al., 2003).

Internationally and in South Africa, the task of assessing the state of the country's biodiversity is hindered by poor quality of available species-distribution data due to sampling bias (Cowling et al., 2001, Van Jaarsveld et al., 1998), currency and scale of data capture (Cowling et al., 2001) with records concentrated around easily accessible or otherwise conveniently surveyed areas, often captured at a coarse spatial scale. In addition, the designation of protected areas has traditionally focused on factors such as perceived aesthetic appeal not in conflict with other forms of land use, not considering biodiversity and ecological processes (Reyers et al., 2001). Owing to the complexity associated with attempting to consider all aspects of biodiversity, proxy measures of biodiversity, known as surrogates, are often used in the place of complete biodiversity data (Faith, 2003, Lawler et al., 2008, Santi et al., 2009).

Vegetation-type, especially in the floristically diverse CFR, has been suggested as an appropriate broad scale surrogate for more general patterns of biodiversity with the underlying assumption that vegetation-type represents a pre-disturbance model of biodiversity in a given area (Mucina et al., 2007). A vegetation-type approach makes comparisons between different management scenarios straightforward as assessing relative representation in a reserve system is simple and inexpensive (Helmer et al., 2002, Santi et al., 2009,). As such, its appeal in conservation planning (Margules & Pressey, 2000) is obvious. Furthermore, vegetation-based surrogates are popular because they are derivable using predictive modelling and remote sensing (Ferrier, 2002, Santi et al., 2009). This approach has commonly been applied in data-poor areas where the surrogate may be the only expansive and reliable biodiversity data available (Mac Nally et al., 2002), allowing comparison across a region of interest without gaps in the data (Santi et al., 2009). Since indigenous fauna, especially large mammals, are inclined to follow the distribution of natural indigenous vegetation (Gaston, 1996, Santi et al., 2009), faunal biodiversity can also be tracked. Vegetation is thus intricately linked to general patterns of biodiversity and theoretically serves as an appropriate proxy by which to judge the state of biodiversity in a given area.

The National Biodiversity Strategy and Action Plan (NBSAP) and the National Biodiversity Framework (NBF) outline priority actions for managing and conserving biodiversity through a systematic biodiversity plan built on earlier conservation plans (Gelderblom et al., 2003, Lochner et al. 2003, Pressey et al., 2003). This includes identification of areas that are still in good ecological condition to consolidate and expand protected areas designed to be spatially efficient and avoiding conflict with other land and water resource uses where possible using the National Protected Area Expansion Strategy (NPAES) (Driver et al., 2012). Considering the richness of biodiversity in South Africa, the relative underrepresentation of many ecosystems in protected areas (Reyers et al., 2001) and the limited resources with which conservation initiatives operate, it is imperative that priority areas for the conservation of specific species and ecosystems be effectively managed through effective information processing and delivery.

With this in mind, the aim of this paper is to investigate the use of vegetation types superimposed over natural vegetation identified from satellite imagery as biodiversity surrogate for conservation planning and management with the goal of prioritizing conservation efforts and facilitating appropriate planning in sensitive areas. In order to achieve this aim, land cover extracted from 1986/7 and 2007 Landsat imagery was studied to establish a baseline from which natural vegetation and land cover change could be identified. The identified remnant of natural vegetation was integrated with vegetation type to enable deductions to be made regarding the implications of land cover changes on biodiversity in the Berg River catchment (Stuckenberg, 2012).

2. Methodology

The Berg River catchment (Figure 1) located in the Berg Water Management Area (WMA) in the Western Cape province of South Africa, drained by the Berg River, is widely regarded as economically and ecologically important due to the rich agricultural areas it supports and the quantity of water it provides to the City of Cape Town. In addition the Berg River catchment has a far higher proportion of critically endangered river ecosystem types than the rest of the country (Driver et al., 2012).

The catchment falls within the winter rainfall regime of the south-western Cape, with annual rainfall generally increasing from the west coast (~300mm) to the mountainous areas (~1400mm) in the eastern portion of the catchment (Schulze, 2007). The geology of the mountainous and upland areas is characterized by quartzites and sandstones of the Cape Supergroup while much of the rest of the catchment is dominated by shales of the Malmesbury and Klipheuwel Groups, and Cape granites (Clark et al., 2007) which have resulted in a catchment typified by nutrient-poor lithologies (De Villiers, 2007).

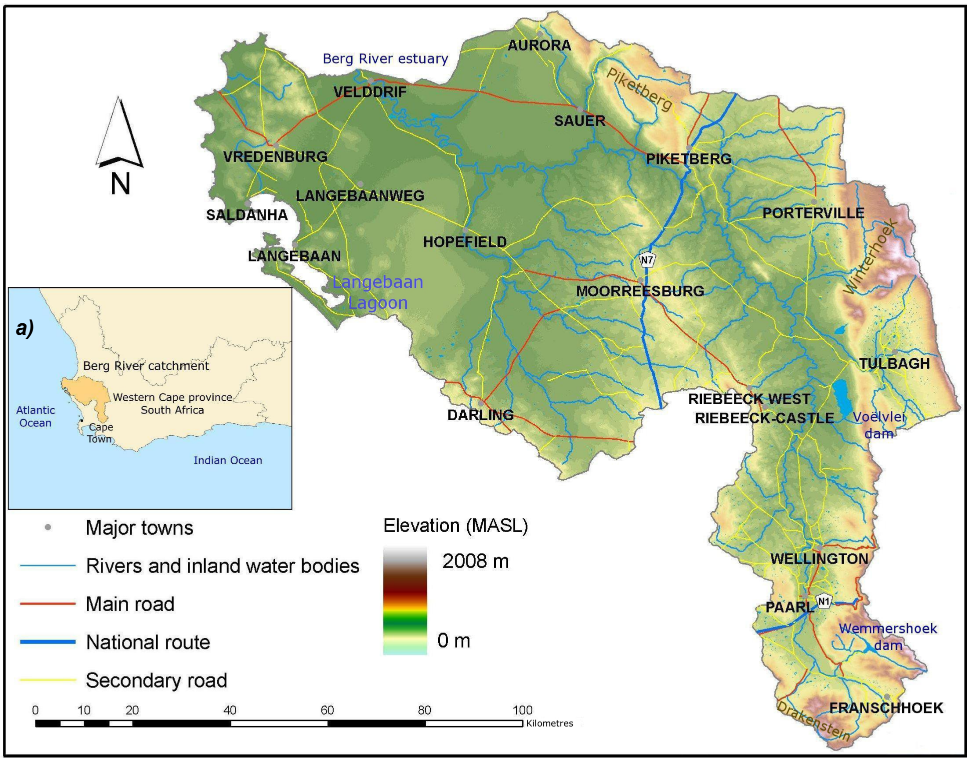

Figure 1. The topography, major towns and major roads of the Berg River catchment. Insert a) indicates study area position relative to SA extent

Soils in this area vary considerably from sandy sediments in the lower catchment to a marked clay accumulation in the middle catchment (Clark et al., 2007). It is the presence of these rich clayey soils which has made much of the catchment appealing to agricultural development and prompted the extensive transformation of large swathes of low-lying land (Kamish, 2008). Shallow, minimally developed soils are common in both the mountainous regions that flank the catchment as well as the low sporadic granite hills that litter the middle catchment (Clark et al., 2007). The soil type predominantly determines the type of vegetation found in the study area (Mucina et al., 2006).

Vegetation in the catchment can broadly be divided into fynbos, renosterveld, strandveld and azonal vegetation types, of which renosterveld is one of the most threatened vegetation types in South Africa. Renosterveld and alluvium fynbos, historically dominating the lowlands of the upper and middle catchment on fertile clays and silts, have been cleared to make way for cultivation with estimates of only 3% of renosterveld remaining. In contrast, fynbos vegetation types, largely confined to fine grained soils at higher elevations, have remained well conserved and have avoided anthropogenic transformation owing to their distribution being unsuitable for agriculture or even urban development. Strandveld and sand fynbos characterizing the lower catchment and coastal areas are usually found on sandy soils with marginal agricultural potential (Mucina et al., 2006).

An inventory was made of all cloud free Landsat TM and ETM+ (Enhanced TM Plus) images in the South African National Space Agency (SANSA) (SANSA, 2013) and United States Geological Survey (USGS) (USGS, 2013) archives covering the Berg River (path/row numbers 175/83 and 176/83). From this inventory it was determined that the earliest cloud free Landsat images covering the study area were captured on 31 December 1986 (scene ID LT517608319861231) and 9 January 1987 (scene ID LT517508319870109). The latest image covering the study area was acquired on 17 February 2007 (scene ID LT517608320070217). All the images were acquired with the TM sensor (Landsat 5). The images have spatial resolutions of 30m and include seven spectral bands (USGS, 2013). The Level 1G images provided by SANSA were orthorectified and projected to the South African Coordinate Reference System (Lo19). Rectified, high resolution (0.5m) aerial photography obtained from the Chief Directorate National Geospatial Information's (CDNGI) and the 5m resolution Stellenbosch University Digital Elevation Model (SUDEM) (Van Niekerk 2012) was used as reference. The images were radiometrically and atmospherically corrected using ATCOR software (Richter & Schläpfer 2011). Multispectral and panchromatic SPOT 5 images (2.5m and 10m resolution respectively) of 2010 were acquired and processed in the same manner as the Landsat imagery.

A revised land cover legend based on the CDNGI land cover legend (Lück, 2006) comprising 10 classes was developed for this study (Table 1). Since the primary concern was to identify the presence of natural vegetation, preferably indigenous, a differentiation between pristine natural vegetation and degraded or otherwise altered vegetation was required. The differentiation was done in accordance with the definition for these classes as described in Lück (2006). Urban vegetated areas were placed into a separate class since grouping features such as golf courses with cultivated land (*Agricultural areas*) might be misleading in a change analysis (Stuckenberg, 2012). For simplicity the class names were abbreviated as shown in Table 1. See Stuckenberg (2012) for detailed description of each class. Henceforth, the abbreviated legend will be used when discussing the results of the mapping exercise.

Two field surveys, from 26/05/2010 to 27/06/2010 and 17/03/2011 to 11/04/2011 respectively, were undertaken to acquire reference data to guide the supervised classification and perform accuracy assessment of the derived land-cover maps. A total of 819 locations were visited, documented and photographed during and before the winter rains to eliminate seasonal differences in land cover and phenological variations in vegetation (Driver et al., 2012). Areas remaining unchanged between the latest set of Landsat image acquisitions (2007) and the field visits (2010/11) were identified on a series of 2010 SPOT5 images (Astrium, 2013) of the study area. Only such areas were included in the field surveys.

Table 1. Modified Land Cover Classification System used to generate land-cover maps of the Berg River catchment

Land-cover class	**Abbreviated legend entry**
Primarily indigenous trees, shrubs, forbs, herbland and graminoids	*Natural vegetation*
Primarily degraded or alien trees, shrubs, forbs, herbland and graminoids	*Semi-natural vegetation*
Natural and semi-natural aquatic or regularly flooded areas (herbaceous)	*Aquatic vegetation*
Natural and semi-natural aquatic or regularly flooded areas (woody)	
Needle-leaved, broadleaved evergreen and broadleaved deciduous trees	*Cultivated trees*
Broadleaved shrubs, herbaceous graminoids and non-graminoids	*Agricultural areas*
Urban vegetated areas	*Urban vegetated areas*
Artificial terrestrial primarily non-vegetated areas	*Artificial bare areas*
Natural terrestrial primarily non-vegetated areas	*Natural bare areas*
Natural or artificial primarily non-vegetated aquatic or regularly flooded water bodies	*Water*

A nearest neighbour (NN) supervised classification was deployed in an object-orientated environment (eCognition Developer 8) (Trimble, 2013) in order to analytically generate two land cover maps (1986/7 and 2007) from the Landsat TM imagery. Supervised classification is an approach commonly used for remotely-sensed data classification (Hubert-Moy et al., 2001; Lu & Weng, 2007) and requires a sufficient number of training samples to perform a successful classification (Mather, 2004; Campbell, 2006; Lu & Weng, 2007; Myburgh & Van Niekerk, 2013). NN is the supervised classifier most commonly employed for object-based supervised classification (Campbell 2006). All TM bands, apart from the thermal infrared band which has a lower resolution, were used as input to the supervised classifier. A bottom-up, region-growing segmentation approach was used to produce consistent results across the relatively large and heterogeneous study area by iteratively minimizing dissimilarity within spatially continuous clusters (regions) and maximizing dissimilarity between regions while merging most similar neighbouring regions (Blaschke & Lang, 2006; Hay & Castilla, 2008; Blaschke, 2010; Fourie, Van Niekerk & Mucina, 2012). Subsequently supervised classification was performed using in excess of 20 training sites for each of the mapped classes (Table 1).

Manual post-classification editing was required for both dates to remove inconsistently identified features such as roads and mountain shadows and to differentiate the class *Natural vegetation* into pristine and degraded natural vegetation. Smaller detected changes resulting from slight incongruences between polygons of the same class were removed and to facilitate the analysis, transformed areas of less than 9 ha (McDonald et al., 1984) were eliminated using a sliver removal technique whereby these were merged with neighbouring polygons of the largest area. Reference data from two field surveys (one during the wet season and one during the dry season) were used in conjunction with data extracted from higher resolution SPOT5 satellite imagery and aerial photography as well as National Land Cover (NLC) data from 2000 (Van Den Berg et al., 2008) and 2009 (SANBI, 2009). A total of 2070 reference points were selected to perform the accuracy assessment on the 2007 land cover map: 100 samples per class were used for each of the 10 classes. Another 1070 sample points were proportionally allocated to each land-cover class based on the area covered by each class. The field campaign contributed 664 of the 2070 sample points, while the remaining 1406 points were selected at random from within various land-cover classes and compared against aerial photographs, captured in 2007, in order to gauge their accuracy. Accuracy was quantified and interpreted using a confusion matrix.

From the classified land-cover maps the class *Natural vegetation* was identified for further processing and integration with vegetation type data and exported as polygon shapefiles. An accuracy assessment could only be carried out on the 2007 land-cover map, because of the lack of available field data for the historical images. Appropriate aerial photography and higher-resolution satellite imagery of the study area were not available to verify the 1986/7 map, however since this map was generated using the same techniques as the 2007 map, it is assumed that it is comparably accurate.

To date three vegetation maps of South Africa have been compiled (Acocks, 1988, Low et al., 1996, Mucina et al., 2007) which delineate areas of relatively homogenous structure, floristic composition and ecological processes into discrete vegetation types (Mucina et al., 2006). For this study the latest of these vegetation maps (Mucina et al., 2007) was selected to act as biodiversity surrogate. In this data set all vegetation types are described in terms of distribution, vegetation and landscape features, geology and soils, climate, biogeographically important and endemic taxa as well as conservation status (Mucina et al., 2006). These vegetation types (also in GIS format) were integrated (intersected) with the areas identified during the land cover mapping as *Natural vegetation*. Change analysis was performed on the integrated data set using IDRISI land change modeller (LCM) (Eastman, 2006).

3. Results

3.1 Land-cover classification

Figure 2 shows the classified land cover maps for 1986/7 and 2007. The accuracy with which most land-cover classes were recorded was fairly good considering the small scale at which they were mapped. The overall accuracy of 2007 land-cover classification was found to be high at just over 85%.

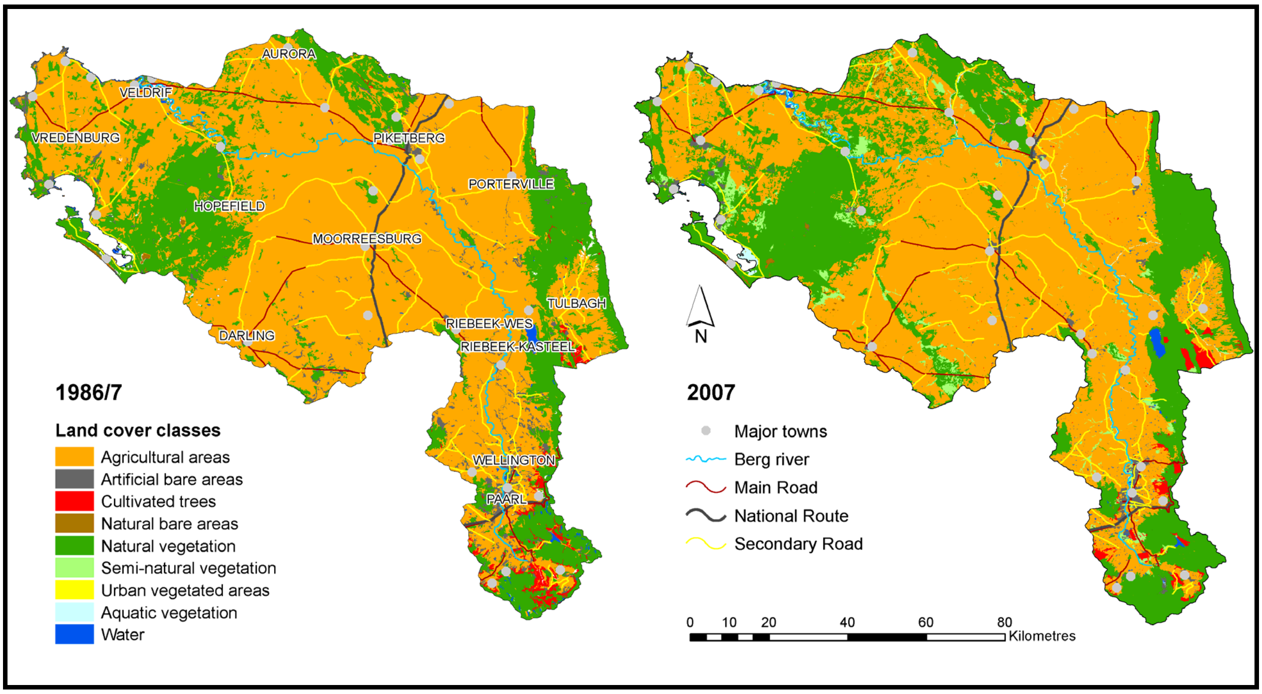

Figure 2. Classified land cover of the Berg River catchment for 1986/7 and 2007

Table 2 provides the confusion matrix with producer's and user's accuracy for the classes of interest. The producer's accuracy relates to the probability that a reference sample will be correctly mapped and measures the errors of omission. Conversely the user's accuracy indicates the probability that a sample from land cover map actually matches what it is from the reference data and measures the error of commission. The column labels under Reference data refer to the number assigned to the land-cover class in the Map data column.

It is clear from Table 2 that *Semi-natural vegetation* (43.8% producer's accuracy) was often confused with *Natural vegetation* and *Agricultural areas*. This confusion was mainly attributed to the inability of Landsat imagery to discriminate between alien and indigenous vegetation. Alien invasive plants often occur in stands that are much smaller than the native 30m spatial resolution of the images and are consequently easily mistaken for natural vegetation. Some abandoned fields in the process of rehabilitation were misclassified as agricultural fields. All the errors that were identified during the accuracy assessment were manually corrected (i.e. objects that were incorrectly classified were manually re-assigned).

Table 2. Confusion matrix, producer's, user's and overall accuracy of selected classes within the 2007 land-cover map

Reference data											
Map data	**(1)**	**(2)**	**(3)**	**(4)**	**(5)**	**(6)**	**(7)**	**(8)**	**(9)**	**Row total**	**Producer's accuracy**
Agricultural areas (1)	696	2	4	0	2	4	51	0	0	759	93.8
Artificial bare areas (2)	2	89	0	9	3	0	12	3	0	118	77.4
Cultivated trees (3)	2	0	103	0	0	0	6	0	0	111	92.8
Natural bare areas (4)	9	6	0	73	15	0	0	0	1	104	88
Natural vegetation (5)	16	1	3	1	336	4	75	0	0	436	91.6
Aquatic vegetation (6)	8	0	0	0	4	185	4	0	6	207	94.9
Semi-natural vegetation (7)	4	0	0	0	4	1	119	0	0	128	43.8
Urban vegetated areas (8)	4	15	0	0	2	0	5	75	0	101	96.2
Water (9)	1	2	1	0	1	1	0	0	100	106	93.5
Column total	742	115	111	83	367	195	272	78	107	2070	
User's accuracy	91.7	75.4	92.8	70.2	77	89.3	93	74.3	94.3		
Overall accuracy	85%										

3.2 Land-cover change analysis

Table 3 shows the areas covered by each of the land-cover classes for the 1986/7 and 2007 land-cover maps. The dominance of cultivation (*Agricultural areas*) in the catchment is evident, constituting approximately 66% (5780km^2) of the total catchment area in 1986/7 and nearly 62% (5489km^2) in 2007. The second-largest land cover in the study area is *Natural vegetation* (~30%). Notable changes in land cover include the expansion of *Artificial bare areas* (25%) and *Urban vegetated areas* (202%) around urban centres. The extent of commercial forestry (*Cultivated trees*) in the upper reaches of the catchment was reduced by 41%. A 5% decrease in *Agricultural areas* was recorded between 1986/7 and 2007, which translates to a reduction of about 290km², much of which occurred in areas of marginal agricultural potential (Borras, 2003, Kirsten et al., 1994).

Caution should be used when interpreting these results as lengthy fallow periods are practiced in the lower catchment, although it is unlikely that this had a significant effect on the total cultivated area (*Agricultural areas*). Much of the vegetation classified as *Semi-natural vegetation* in 1986/7 was classified as *Natural vegetation* in 2007, leading to an increase in this class of 14% over 20 years (Table 3). Figure 3 illustrates the conversion to and from *Natural vegetation* in the catchment. The next section relates this result with vegetation types to gain a better understanding of how biodiversity was impacted.

Table 3. Overview of land-cover changes in the Berg River catchment

Land-cover class	1986/7		2007		1986/7 – 2007
	Area (km²)	%	Area (km²)	%	%
Agricultural areas	5780	65.9	5489	61.6	-5.0
Artificial bare areas	117	1.3	147	1.7	25.3
Cultivated trees	151	1.7	88	1.0	-41.2
Natural bare areas	33	0.4	35	0.4	6.1
Natural vegetation	2445	27.5	2797	31.4	14.4
Semi-natural vegetation	276	3.1	232	2.6	-15.7
Urban vegetated areas	2	0.02	6	0.07	202.9

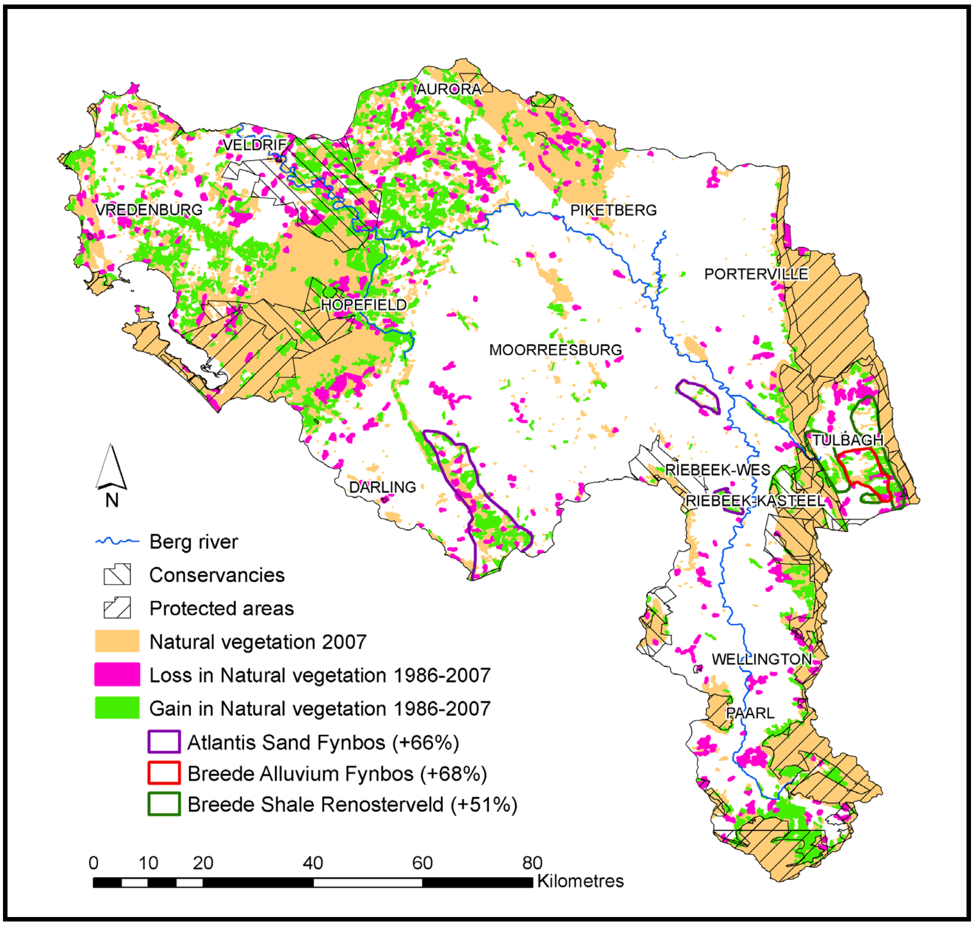

Figure 3. Gains and losses in natural vegetation from land cover change analysis indicating protected areas and conservancies in the Berg River catchment

3.3 Natural vegetation and vegetation types analysis

Table 4 summarises the results of the land cover change analysis when integrated with the vegetation types. The table includes the extent of the particular vegetation type that occurs in the catchment (labelled potential extent) as well as percentage change during the study period. Only vegetation types that cover greater than 60% of the potential extent of a particular vegetation type in the 1986/7 land-cover map are considered. Vegetation types occupying relatively small areas are more prone to drastic changes since small alterations to their extent contributed to seemingly significant overall changes. Examples include *Atlantis Sand Fynbos* (+66%), *Breede Alluvium Fynbos* (+68%), *Breede Shale Renosterveld* (+51%) and *Cape Seashore Vegetation* (+111%) of which *Atlantis Sand Fynbos* and *Breede Shale Renosterveld* are regarded as vulnerable, while *Breede Alluvium Fynbos* is classified as endangered with a very low protection level. The greatest loss in vegetation type occurred in the *Leipoldtville Sand Fynbos* (-45%) of which only 11% occurs within this study area while the *Swartland Granite Renosterveld* reduced by 1.1 km^2 to decline by 14%. Especially loss of natural vegetation within conservancies and protected areas are of concern and needs to be addressed.

Table 4. Changes in vegetation types in the Berg River catchment

Vegetation type	**Percentage of potential extent 1986/7**	**Percentage change**
		1986/7–2007
Cape Winelands Shale Fynbos	60.8	11.8
Hawequas Sandstone Fynbos	87.9	3.6
Kogelberg Sandstone Fynbos	83.9	13.6
Olifants Sandstone Fynbos	97.6	-0.2
Piketberg Sandstone Fynbos	79.9	5.0
Western Altimontane Sandstone Fynbos	100.0	0.0
Winterhoek Sandstone Fynbos	99.3	-0.7
Western Coastal Shale Band Vegetation	81.7	14.6
Northern Inland Shale Band Vegetation	92.1	1.5
Cape Estuarine Salt Marshes	69.9	2.8
Southern Afrotemperate Forest	95.0	5.2

The most important gains were to *Western Coastal Shale Band Vegetation*, *Kogelberg Sandstone Fynbos* and *Cape Winelands Shale Fynbos*. Small losses occurred in *Olifants Sandstone Fynbos* and *Winterhoek Sandstone Fynbos*.

4. Discussion

The notable changes in *Artificial bare areas* and *Urban vegetated areas* around urban centres (Table 3) can be ascribed to accelerated urban expansion and golf course developments. The loss of *Cultivated trees* (commercial forestry) is in part due to the lack of long-term profitability and high rate of water consumption (Ruiz, 2003), as well as forest fires destroying commercial forest plantations around Franschhoek (Currie et al., 2009). In addition, the Working for Water (WfW) programme has attempted to remove remaining *pinus* specimens as well as other invaders, particularly in riparian areas (Currie et al., 2009). Unfortunately *Natural vegetation* reclaimed from previously cultivated areas (*Agricultural areas, Cultivated trees*) is unlikely to exhibit as rich a compositional diversity as do areas of pristine vegetation cover. While rehabilitation programs could make considerable advances to recovering original biodiversity patterns, cash-strapped conservation agencies just do not have the resources needed to rehabilitate these areas (Kraaij et al., 2011).

Most fynbos vegetation types in the Berg River catchment are located in mountainous areas often within mountain catchment reserves which have limited potential for agriculture or settlement, have been spared extensive transformation, making significant transformations unlikely in the near future. Therefore most fynbos vegetation remained fairly static over the study period, the exception being a few fynbos types in low-lying areas. The increase shown in *Breede Alluvium Fynbos*, located within the Tulbagh valley is attributable to the abandonment of several agricultural fields over time. Much of the Renosterveld vegetation type, typically found on flat clayey soils well-suited for several types of cultivation, had already been transformed prior to the earliest satellite images used in this study. Encouragingly, gains in the area occupied by most Renosterveld vegetation types were recorded in the study. This corroborates findings (Heelemann, 2010) which note that Renosterveld will spontaneously re-establish to some degree but will seldom display extensive species diversity, even several decades after the cessation of agricultural activity (Memiaghe, 2008). *Elytropappus rhinocerotis* (Renosterbos) is known to re-establish fairly quickly giving the superficial impression of an indigenous vegetation cover. Kemper, Cowling & Richardson (1999) found species diversity in even very small renosterveld fragments located in agricultural areas to be high, strengthening the case for preservation of remaining patches by private landowners through the Biodiversity and Wine initiative (www.wwf.org.za). A more detailed study is required to suggest ways of restoring connectivity between such fragments. Many of the Sand fynbos and Strandveld vegetation types are located on sandy soils in areas of marginal agricultural potential and as a result are only partially transformed. The 66% increase noted in the vulnerable *Atlantis Sand Fynbos* is attributable to the establishment of the Riverlands Nature Reserve where considerable effort was placed into restoring indigenous vegetation with varying levels of success (Holmes, 2008). However, the analysis of reference data confirms that the expansion of invasive species (wattles, invasive annual and perennial grasses such as *Hyparrhenia*) lend these areas an impression of regeneration borne out by the low producer's accuracy of 27.6% for *Semi-natural vegetation* vs. *Natural vegetation* (Table 2).

Noteworthy gains (Table 4) in most vegetation types have been demonstrated in the Berg River catchment by this study. While this is a positive finding, concern has been raised over the degree to which newly identified natural vegetation can be taken to represent the diversity in species composition and structure for which many of these vegetation types are so well known since it is unlikely to register these differences using medium or coarse resolution satellite imagery. This is a major obstacle for this type of approach where the potentially significant changes in indigenous biodiversity cannot be adequately measured as was evidenced by large areas of *Natural vegetation* types displaying a level of species diversity incomparable to a pristine vegetation type. However, since restoration initiatives can achieve benefits to both biodiversity and provision of ecosystem services (Bullock et al., 2011), the importance of the land cover-vegetation type approach cannot be underestimated in highlighting areas where restoration initiatives may achieve measurable results.

In the CFR a means of extracting a measure of the compositional diversity of different vegetation communities from remotely sensed data would greatly enhance the capacity of remotely-sensed data to monitor biodiversity. Owing to the plant heterogeneity within vegetation types and the superficial similarity between degraded or invaded areas, it is unlikely that automated spectral classification or even visual interpretation of medium to high resolution satellite imagery (e.g. Landsat and SPOT) will effectively illuminate vegetation composition. However, the resent availability of very high resolution (0.5m) satellite and aerial imagery may improve the identification of alien plants and help differentiate between pristine and degraded indigenous vegetation (Rouget et al., 2003). More research is required to investigate the use of very high resolution imagery for biodiversity monitoring.

In this instance the extent or change in occupancy of different vegetation types should be seen as a first step in a holistic assessment of biodiversity, in support of National Biodiversity Assessment guidelines for assessment of ecological conditions (Driver et al., 2012). The next step would be to carry out more detailed assessments of factors such as species composition and the integrity of ecological processes. A useful aspect of the study is its ability to identify areas where biodiversity changes are likely to be taking place. In this way an approach that assesses changes in potential vegetation with land-cover maps can direct focused research in an efficient manner. To adequately assess the impact of land-cover changes on biodiversity in the Berg River catchment it is necessary to conduct further field research focused on establishing the health of patches of natural vegetation experiencing change. In this way land cover derived for satellite imagery can be used as the first stage in a comprehensive biodiversity assessment, using the breadth and expedience of remote sensing to hone more detailed and time consuming field surveys.

In this paper we made use of Landsat imagery to demonstrate the efficacy of remote sensing for long-term biodiversity monitoring. The methodology can, however, also be applied to other types of imagery. For instance, SPOT (Système Pour l'Observation de la Terre) and Indian Remote Sensing (IRS) imagery has been available since 1986 and 1988 respectively.

5. Conclusion

The aim of this paper was to investigate the use of vegetation types superimposed over natural vegetation identified from satellite imagery as biodiversity surrogate for conservation planning and management with the goal of prioritizing conservation efforts and facilitating appropriate planning in sensitive areas. The study found that the Berg River catchment experienced a significant increase in natural vegetation from 1987 to 2007. While this is a positive finding, conclusions about the level of biodiversity in these gained areas could not be drawn. The research did, however, demonstrate how remote sensing and GIS can be used as a scoping mechanism for identifying areas where more detailed studies are required, thereby highlighting the potential of remote sensing to monitor biodiversity in the CFR. Landsat data was found to be very useful for establishing an overview of biodiversity change over a long (e.g. 30-year) period, but more research is needed to investigate the efficacy of higher resolution satellite and aerial imagery for carrying out more detailed analyses over shorter (e.g. 5-year) periods. There is an urgent need to develop more accurate and comprehensive monitoring systems for biodiversity in South Africa and especially in the CFR. Large gaps exist in our capacity to translate changes in land cover into changes in biodiversity at all levels of biological organization and to devise conservation and management plans in the light of these impacts.

6. Acknowledgement

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