Spatio-temporal dynamics of land-cover change in a multi-use conservation landscape: The case of the Kruger to Canyons Biosphere Reserve, South Africa

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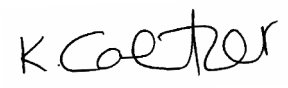
Declaration

I declare that this thesis is my own, unaided work, unless otherwise specified within the text. It is being submitted for the Degree of Doctor of Philosophy at the University of the Witwatersrand, Johannesburg, South Africa. It has not been submitted before for any other degree or examination at any other university.



Kaera Leigh Coetzer

24th day of **March 2014** in Johannesburg

“I may not have gone where I intended to go, but I think I have ended up where I needed to be.”

Douglas Adams, *The Long Dark Tea-Time of the Soul*

Abstract

The Biosphere Reserve (BR) model of UNESCO's Man and the Biosphere Programme (MAB) reflects a shift towards more socially accountable conservation, setting a basis for improving the relationships between people and their environment. The BR approach of dual 'conservation' and 'sustainable development' objectives is a long-term perspective that increases people's ability to sustainably manage environmental resources into the future. By 2013 the MAB Programme had resulted in 621 sites designated globally since first inception in 1976, with 11 of these new listings since 2012. Yet regardless of this global spatial expansion, there are few examples of BRs successfully conforming to the MAB's full criteria, as the reality of aligning conservation and development goals in this model has typically favoured one over the other, with examples of development objectives fuelling broad-scale landscape transformation and conservation objectives that have ignored socio-economic opportunities. However there are a number of case studies presented herein that have shown considerable success, and this thesis suggests that the potential opportunities inherent in the model may outweigh implementation challenges. This thesis critically reviews the history and evolution of the Programme internationally, and makes recommendations for the future of the MAB Programme.

In the South African context, the BR model has been applied to a regional conservation landscape in the north-eastern part of the country. The Kruger to Canyons Biosphere Reserve (K2C) is a long-established land-use mosaic of socio-economic juxtapositions, where nationally important economic sectors and world-renowned protected areas (e.g. the Kruger National Park) neighbour extensive, mostly rural, human populations engaged in informal livelihood practices. These vast settlement expanses are artefacts of the former political system in South Africa, and have been prioritised by the South African National Government for urgent socio-economic upliftment.

Given that understanding the underlying land-use mosaic is essential for successful implementation of the BR model in any landscape, this thesis assessed land-cover change across the BR (1993 – 2006 – 2012), through the creation of land-cover classification maps, analysing spatial changes in cover relative to the MAB theoretical aspirations. It focuses on three consolidated land-cover cover classes: Intact Vegetation, Impacted Vegetation and Settlement,

that present a graduated scale of land-use intensities across the BR, acting as surrogates for the ‘environmental’ and ‘human utilisation’ aspects of the BR approach.

Results indicate that the spatial distribution of land-cover change in K2C bears little regard for the theoretical BR zonation and the relative limitations on ‘use’ that typifies successful BR implementation. The spatial expansion of Settlement cover replaced adjacent Impacted Vegetation, which in turn encroached in original areas of Intact Vegetation cover, increasingly fragmenting Intact Vegetation across K2C. The expansion of Impacted Vegetation has resulted in the spatial consolidation of communal use areas, which has likely serious implications for the long-term sustainability of current harvesting levels. Anecdotal field observations have indicated community responses to these concerns with the formal fencing of rangelands to secure ownership and control access.

Losses of Intact vegetation translate spatially into an area >400km^2, with observed land-cover changes occurring beyond the transition zone, encroaching into the (fully-protected) core and (partially–protected) buffer zones. The increased rate of landscape change in the recent observation period (2.3%$_{1993\text{-}2006}$ versus 5.7%$_{2006\text{-}2012}$) poses challenges for landscape management, with future predictions of escalating transformation likely to undermine BR sustainability, as well as options for a regionally connected protected area network. Historical landscape trends predict a further 5.1% loss of intact vegetation in the next 6 years (2012 – 2018), yet this may over-estimate the potential for transformation in this landscape. However the possibility of a similar loss over a longer temporal period, i.e. 12 years, is not improbable, with the unprotected matrix landscape of the transition zone having experienced comparable losses since 1993, i.e. 4.17%.

K2C has had substantial success with pilot socio-economic projects, and has received formal international recognition for the stakeholder engagement / alignment occurring in the subregion as a result of the BR’s listing here. Given these successes, and the considerable investment in its establishment, it is likely that the BR model here will endure. The results of this thesis contribute to improving implementation success long-term, informing policy and management priorities across the BR. These include contributions to *i)* a marketing approach for the BR, specifically the importance of a K2C ‘brand’ for enhancing co-operation and strengthening the BR’s regional identity, as well as *ii)* providing support for land-use planning and decisions related to land-use practices and BR sustainability.

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Disclaimer

This thesis consists of a sequence of content chapters that have been prepared for publication in a range of scientific journals. This has meant that formatting styles vary between chapters, with some degree of overlap to ensure publishable papers.

Author contributions, areas of overlap and specific Journals have been specified in Chapter 1.

Table of Contents

List of Figures

Chapter 1

Chapter 2

Chapter 3

Chapter 5

List of Tables

List of Appendices

Chapter 5

List of Abbreviations

AI	Aggregation index
AWMPS	Area weighted mean patch size
AWMSI	Area weighted mean shape index
BR	Biosphere Reserve
BNR	Bushbuckridge Nature Reserve
GTP	Ground-truth points
ICDP	Integrated Conservation and Development Project
IDH	Intermediate disturbance hypothesis
IJI	Interspersion and Juxtaposition
K2C	Kruger to Canyons
KNP	Kruger National Park
LULCC	Land-use / Land-cover change
MAB	Man and the Biosphere
NLC	National land-cover
NP	Number of patches
PA	Protected Area
PLAND	Percentage contribution to total landscape area
SES	Socio-ecological system
SLC	Scan Line Corrector
UNESCO	United Nations Scientific and Cultural Organisation
USGS	United States Geological Survey
WELD	Web-Enabled Landsat Data
WNBR	World Network of Biosphere Reserves

Chapter 1

General Introduction: Conservation in a multi-use landscape

General Introduction: Conservation in a multi-use landscape

1.1 Study Rationale

The Kruger to Canyons Biosphere Reserve (K2C, www.kruger2canyons.com) represents a clearly defined socio-ecological system and a multi-use landscape. Established in 2001 by the Man and the Biosphere (MAB) programme of UNESCO, it is the second largest Biosphere Reserve (BR) in SA, and one of the largest in the world. Through the establishment of BRs, MAB sets a basis for improving the relationships between people and their environment (Bridgewater, 2002), a long-term approach that increases people's ability to sustainably manage environmental resources into the future.

In K2C, the conservation land-use dominates spatially, largely due to the inclusion of the central portion of the Kruger National Park in the BR. Yet the portions of the subregion, i.e. specifically grassland and escarpment areas, have been highlighted by national and regional conservation planning assessments (Driver et al., 2005; Ferrar and Lötter, 2007) as requiring further conservation action, owing to a national history of *ad hoc* protected area placement (Rouget et al., 2003a), and rapid land-cover change in this landscape.

Ecosystems are contiguous, and unrestricted to formal PA fences, and as a result, management of the entire landscape mosaic is required for successful, and enduring, conservation and the maintenance of ecosystem function (Shreeve and Dennis, 2011). In this context, the matrix of land-use and remaining fragments of intact habitat outside K2C's existing PAs are important not only for effective formal biodiversity conservation, but also for the production industries, e.g. forestry, agriculture (Fischer et al., 2006), and for the sustainability of rural livelihoods (Chazdon et al., 2009). Similarly, under climate change scenarios, species responses may further reduce the effectiveness of the existing PAs, requiring that PAs are linked, enlarged, or alternative conservation strategies identified to ensure a functionally connected regional conservation landscape (Beier and Brost, 2010).

Thus, off-reserve conservation and matrix management has become an imperative. Yet the future of land-use/land-cover change (LULCC) within the unprotected matrix may result in an increasingly isolated PA network, potentially undermining the success of off-reserve conservation initiatives, and the longevity and integrity, of matrix biodiversity. The K2C landscape mosaic is one of diverse, and potentially conflicting land-use objectives, with strict environmental protection, commercial agriculture, plantation forestry, mining, and a growing, mostly rural population, juxtaposed across the BR landscape.

With this in mind, there is a need to understand the processes driving land-cover change in the matrix, and the implications for extant biodiversity and existing conservation values. The sustainability of this social-ecological system will depend on its resilience and vulnerability to adverse change (Turner, 2010; Derissen et al., 2011), which in turn will depend on the success of these social, environmental and economic relationships in maintaining social and ecological diversity (Bridgewater, 2002).

Pool-Stanvliet (2013) argues that BRs are models of ‘sustainable development in action’. While this may in theory characterise the MAB approach, should the BR model not deliver on its promise long-term, in terms of both conservation and development opportunities, it may be that an alternative management strategy is required for the South African context. As a result, in K2C, the dynamic interrelations between the intact ecosystems, environmental utilisation (at multiple scales of measurement and intensity of utilisation), and resultant land-cover change are a priority for research.

At this point, there are no other studies that have addressed this at the scale of the BR, a trans-boundary conservation designation that traverses provincial and PA managerial boundaries.

I argue that this is a research gap that urgently needs to be fulfilled given the following interrelated reasons:

- *i)* the potential for spatial conflicts between the local human BR population, existing land-use types and the ‘conservation’ land-use (e.g. Rouget et al., 2003b; Ferrar and Lötter, 2007; Naidoo and Ricketts, 2006);
- *ii)* the interdependencies between changing socio-economic conditions, more extensive cf. intensive resource utilisation, and further fragmentation of intact matrix vegetation cover over time (e.g. Lambin and Meyfroidt, 2010);
- *iii)* the links between landscape connectivity and climate change scenarios (e.g. Opdam and Wascher, 2004; Heller and Zavaleta, 2009); and

iv) the links between environmental utilisation, the extent and intensity of LULCC, and K2C's long-term functioning as a 'Biosphere Reserve' and sustainable conservation landscape (e.g. Ma et al., 2009).

This project aims to address these issues, in a manner intended to facilitate the translation of research results into usable products relevant for utilisation by BR decision-makers. Research offers little value if the end-products remain purely academic and cannot be applied into practical solutions for real-world challenges (e.g. Reyers et al., 2007).

1.2 Background literature

1.2.1 Conservation beyond the individual Protected Area

Formal protected areas (PAs) now conserve ~13% of the global land surface (Buchart et al., 2012). This is a substantial increase since Zimmerer et al. (2004)'s estimation of 3.47% in 1985, and most likely to increase further with the global Aichi Biodiversity Targets (Target 11) setting this to 17% by 2020 (www.cbd.int/sp/targets/; Jenkins and Jobba, 2009). However, there is increasing awareness that there are large gaps in protected area (PA) coverage at global and regional scales (Driver et al., 2005; Naughton-Treves et al., 2005), which are likely to be further compromised under future climate change scenarios (Hannah et al., 2007).
Recent literature has emphasised the connectedness of ecological processes acting at scales larger than the individual PA (Persha et al., 2010), and for PAs to continue to be effective, they cannot be managed in isolation from the landscapes in which they are embedded (Hansen and DeFries, 2007; Brady et al., 2009). As such, landscape / regional approaches to conservation have become important (Ashley et al., 2006; Wiens et al., 2009; Rands et al., 2010), with the matrix land-cover beyond the PA offering both conservation challenges (e.g. increased isolation, altering microclimates, changing disturbance regimes and local invasion potentials), as well as potential opportunities (e.g. additional resource provisioning, facilitating dispersal, and increased landscape heterogeneity; reviewed in Driscoll et al., 2013), with the degree of fragmentation and the levels of human utilisation important for determining ecological responses (Hanski, 2011).

Landscape approaches to conservation necessitate that matrix land-users are included within the conservation decision-making arena (Sayer et al., 2013). Yet, in human-dominated conservation landscapes there is often the notion that the contemporary role of PAs must go beyond only the conservation of biodiversity, and include direct benefits to human livelihoods as part of their functioning (Salasfsky and Wollenberg, 2000; Adams et al, 2004; Naughton-Treves et al., 2005). The trend towards coupled conservation and socio-economic development outcomes is largely founded on the ideas of social equity and power relations (Redpath et al., 2013), with the historical displacements (Agrawal and Redford, 2009) and the lost-opportunity costs for livelihoods / other land-users that are typical of exclusionary / protectionist approaches driving the discussion. However, there remains considerable debate around the practicality of aligning of environmental protection and poverty alleviation strategies (Miller et al., 2011; Minteer and Miller, 2011; Rantala et al., 2013), with trade-offs likely in reconciling what are traditionally, disparate policy objectives (Adams et al., 2004; McShane et al., 2011; Miller et al., 2012).

1.2.2 Conservation in human-dominated landscapes: The Biosphere Reserve Approach

UNESCO's Man and the Biosphere (MAB) Programme reflects the shift towards both people-centred and landscape conservation approaches, with the Biosphere Reserve (BR) designation explicitly incorporating humans, and sustainable development goals alongside ecosystem protection (UNESCO, 1996).

BRs are not intended to displace traditional PAs from their role in conserving biodiversity. Rather, they are expected to facilitate co-operation across administrative boundaries, and enable socio-economic development, whilst maintaining existing conservation values through their structured spatial approach to environmental utilisation (Pool-Stanvliet, 2013; also Coetzer et al., 2013a: Chapter 2 of this document).

The programme has had considerable spatial expansion internationally since the first set of 58 BRs were designated in 1976 (as collated in Coetzer et al., 2013a), and as of 2013, 621 BRs have been listed in over 117 countries (http://www.unesco.org). However, as with other people-conservation approaches (see Agrawal and Redford, 2006; Miller et al., 2012), the successes and limitations of the BR model have been poorly documented in the formal literature. As a result, I have provided a critical review of the MAB Programme, its evolution, *status quo*, and associated guiding principles in Chapter 2.

To avoid repetition, this has not been addressed further in this section.

In the South African context, the first BR to be listed with MAB, Kogelberg BR, was designated in 1998 in the Western Cape region (www.kogelbergbiospherereserve.co.za/). With the designation of this BR, the Western Cape provincial authorities identified BRs as the preferred spatial model for bioregional planning, with development and management frameworks to be aligned with BRs spatial zonation (Pool-Stanvliet, 2013). Subsequently a further 5 BRs have been established in South Africa from 2000 – 2009 (http://www.unesco.org; Fig. 1), with a spatial distribution concentrated in the south-west and north-eastern regions of the country. Together existing BRs have a spatial coverage of more than 6 300 000 ha.

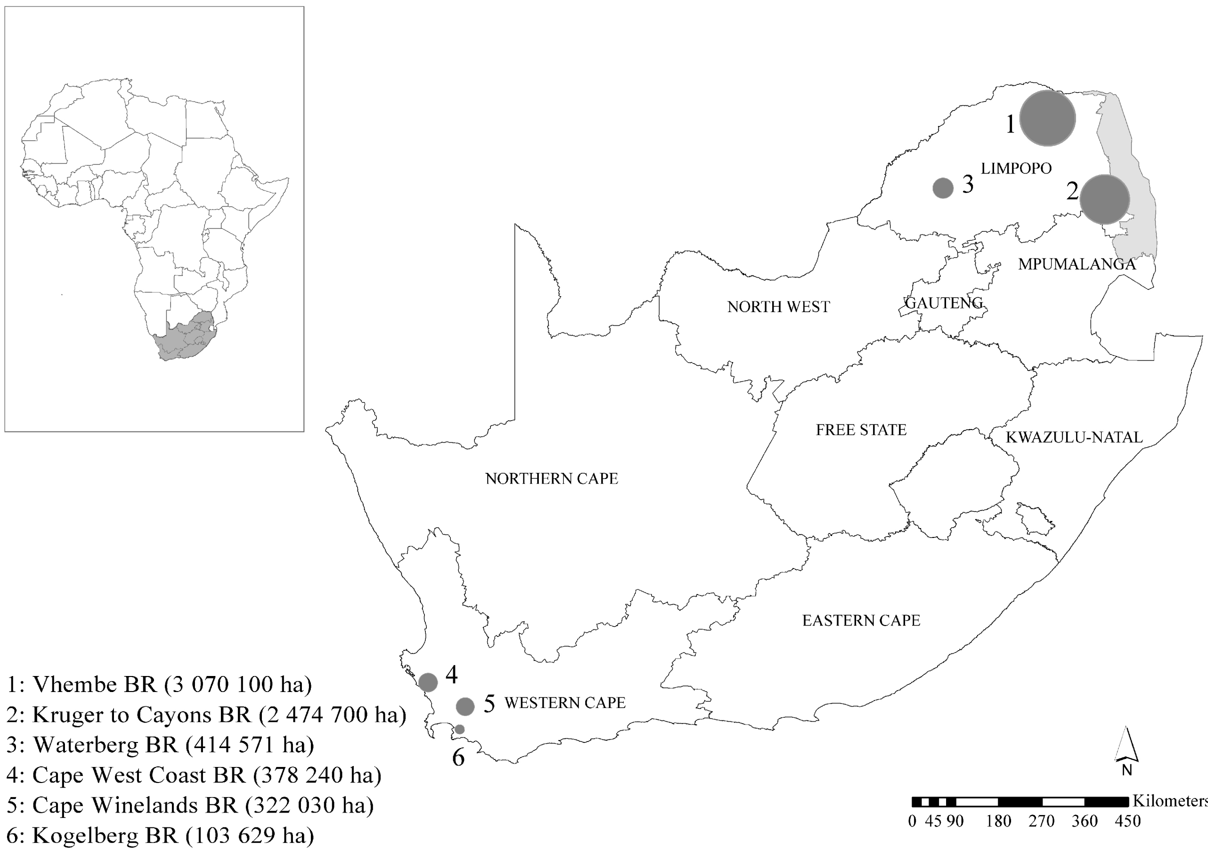

Figure 1. Spatial distribution of existing Biosphere Reserves (BR) across South Africa, including provincial boundaries and continental position (map inset). Point locations of each BR rather than the accurate boundaries of each BR are indicated, with markers illustrating the relative proportional size of each site. Actual sizes (in hectares) are listed with the legend key.

Regardless of the increasing 'buy-in' to the Programme in the country, in general BRs have received limited support from the South African National Government. While the Western Cape provincial government may set the precedence for formalising BR specific legislation, with the *Biosphere Reserves Act* (Act 6 of 2011) providing support for designation and management of BRs in the province, at a national scale, BRs in SA are managed as non-profit organisations, each responsible for securing funding and ensuring the successful management of the designation (Pool-Stanvliet, 2013). This raises challenges for practical implementation, both in

South Africa and internationally where similar circumstances prevail. Frequently, the BR concept remains poorly known, often misunderstood, and sometimes viewed as in direct opposition to biodiversity conservation (Coetzer et al., 2013a).

Pool-Stanvliet (2013) argues that the effectiveness of the MAB Programme in South Africa will be measured by the ability of established BRs to act as regional testing grounds for sustainable land-use management. In this regard, the underlying socio-economic context will become a fundamental determinant of success. For those BRs, faced with high population growth rates, entrenched poverty, and high levels of unemployment, e.g. Kruger to Canyons BR (www.kruger2canyons.org/), land-use decision making performed at local scales (individual, household, community) is likely to be a significant factor in determining the sustainability of regional land-use management practices.

The Kruger to Canyons BR (K2C) has an (initially rural, now densifying) population of ~1.5 million people, many of whom still reside in areas delineated as the former 'homeland' regions under the Apartheid segregationist system. In these areas there is a heavy reliance on state provided social grants, i.e. pensions, disability or child grants, and/or migrant remittances from family members employed in urban centres (e.g. Shackleton & Shackleton, 2002; Madubansi & Shackleton, 2006; Cousins et al., 2007); providing crucial sources of cash income for many households. The majority of households are engaged in subsistence livelihood practices, harvesting directly from for the local environment (i.e. communal rangeland areas) to meet daily food, fuel and financial needs (Twine, 2005; Makhado et al., 2009). The poor opportunities for formal employment in the subregion have ensured that communities frequently derive more benefit from subsistence strategies than sources of cash income long-term (Dovie et al., 2005), with the direct-use financial benefits of harvested resources for communities living within the subregion ranging from approximately \$400 - \$700 per user/household/annum (Dovie et al., 2002; Dovie et al., 2005; Dovie et al., 2006), with the indirect use value ('savings value') of land-based strategies typically under-estimated in financial assessments (Shackleton et al., 2001).

Given the importance of harvested resources for ensuring livelihoods in K2C, there have been concerns around the sustainability of resource extraction. Banks et al. (1996) predicted large-scale and complete deforestation around villages in the southern part of the BR within a 15 year period. Recent re-evaluation of the same areas have indicated that deforestation has not occurred to the extent as originally anticipated (e.g. Fisher et al., 2012; Matsika et al., 2012), but there are indications that extraction is becoming unsustainable - even with the improvements made to electricity provisioning here in the last two decades (Madubansi and Shackleton, 2006). Changes in consumption and extraction patterns as resources become increasingly more difficult to locate, or as grant / remittance flows fluctuate over time (with impacts on harvesting intensities, i.e. Campbell et al., 2000), will add further complexity to BR management, and likely become a priority in ensuring the success of the model here long-term.

1.2.3 The Role of Remote Sensing in Monitoring Landscapes

There is a global consensus that land-cover changes present one of the greatest threats to biodiversity persistence (Sala et al., 2000; Reyers et al., 2002), with as much as half of the ice-free land surface transformed by anthropogenic forcing (Chapin et al., 2000; Turner et al., 2007). As a result, all signatories to the Convention on Biological Diversity are required to assess the impact on land transformation on biodiversity, and where necessary initiate the appropriate responses (Reyers et al., 2002).

Remote-sensing has become an effective tool for monitoring biodiversity features, characterizing the ‘signature’ (e.g. the extent and intensity) of human resource use across the landscape, and informing conservation management and policy decisions over time (for review, Kerr and Ostrovsky, 2003; Rosenquist et al., 2003; Turner et al., 2003; Cohen and Goward, 2006; Gillespie et al., 2008; Kennedy et al., 2009; Wang et al., 2010; Nagendra et al., 2013). Understanding the landscape’s history of transformation may reveal unexpected drivers of the change (Bender et al., 2005), providing awareness into the abiotic / biotic/ socio-economic / political relationships in the landscape - all of which are useful in quantifying current and future predictions of environmental threats (e.g. Ried et al., 1999).

However, there has been criticism of a remotely-sensed approach to environmental management as ‘armchair environmental classification’ (Brooks et al., 2004), lacking the sensitivity of species data, and arbitrarily homogenising and delineating land-cover / land-use with little consistency across classification schemes (Brooks et al., 2004; Thogmartin et al., 2004;Verburg et al., 2011; Fisher et al., 2013). Inherent uncertainties and vagueness exist within remotely-sensed data (Benz et al., 2004), i.e. noise within the sensor, uncertainties in the image generation process and fuzzy concepts within classification schemes (e.g. delineation of vegetation boundaries), all of which affect the accurate representation of the real-world. Concerns linked to the levels of disagreement between derived cover and reference data has affected the applicability of land-cover maps, with the historical lack of error reporting undermining confidence in and the likely utility of classification maps for practical application (Foody, 2002).

Yet for large spatial areas, inaccessible areas, or instances where historical landscape understanding is required, remotely sensed data is often the only practical option available (Townsend et al., 2009). Technical advances in data processing and analysis methodologies (Cihlar, 2000), the reduction in remotely-sensed data acquisition costs (with a significant portion of medium resolution archived data sets now freely available via Internet download, e.g. Landsat, SPOT, MODIS; see Roy et al., 2011), and improved spatial and temporal coverage have substantially enhanced the usability of remotely-sensed data for ecological purposes (Cihlar, 2000; Coppin, 2004). Recent advances in sensor capabilities have improved the characterisation of fine-scale landscape heterogeneity, tracking the dynamics of individual species, or measurements of ecosystem health, and complementing broad-scale / regional land-cover assessments (e.g. Asner et al., 2004; Spanhove et al., 2012; Fisher et al., 2013). Improvements made in capturing spatial data-sets, e.g. transport networks or population densities, have greatly enhanced the ability of remotely-sensed data to identify the drivers of land-cover change, and provide 'stressor / pressure' assessments of transformation across the landscape (Wang et al., 2009; Nagendra et al., 2013). Similarly, the progress made in end-user manipulation of remotely-sensed data, and the trend towards increased reporting of potential errors, assumptions made, or levels of 'mismatch' (Foody, 2002), has increased the utility of all remotely-sensed data for research and management, regardless of sensor resolution.

1.2.4 Understanding land-cover change

In recent decades quantifying and understanding the patterns and processes of land-use/ land-cover change (LULCC) has become a global priority, with landscape changes affecting (eco)system outcomes, e.g. resilience and sustainability issues, and the vulnerability of human-environmental relationships (Turner, 2007; Aldwaik and Pontius, 2012). Land-cover changes are responses to complex social and biophysical drivers operating at multiple spatial and temporal scales (Brown et al., 2004; Turner and Robbins, 2008), with feedbacks between landscape features, human decision-making and associated contextual circumstances important for understanding the causality of transformation (Geist and Lambin, 2001; Verburg, 2006; Perz and Almeyada, 2010). The complexity of these relationships, and thus, the dynamics of LULCC have ensured that causal explanation has become interdisciplinary in nature, drawing on both the social and natural sciences for interpretation (Brown et al., 2004).

From a landscape ecology perspective, Skole et al. (1994) and Lambin (1997) identified key analyses that enable understanding of the processes of land-cover change. These, listed below (as per the representation of Petit et al. (2001)), describe what could be arguably considered the typical strategy for determining the spatial behaviour of LULCC, combining quantitative and qualitative information to quantify changes and predict the future consequence of these changes for a particular landscape (see Aldwaik and Pontius, 2012).

1. *Explanatory analysis*:

This includes the identification of the main ecological and socio-economic variables driving land-cover change (Petit et al., 2001). Case studies and field observations provide insight into local scale dynamics of LULCC, i.e. ecological and social contextual circumstances, which may in turn have larger-scale applicability (Skole et al., 1994; Lambin, 1997). However, in large and/or heterogeneous regions, the drivers of change are likely to differ across the landscape. This adds spatial complexity to any attempt at causal explanation of land-cover change (Brown et al., 2004), as natural resource management tends to be site-specific, and thus extrapolating local decision-making to larger scales is likely to be challenging (Castella et al., 2007). Further complicating explanatory analyses are the interactions and feedbacks between the causal agents of land-cover change, and the scale at which these occur (Geist and Lambin, 2001; Lambin et al., 2003; Brown et al., 2004). For example, local-level LULCC is frequently influenced by exogenous forces, originating from macro-level decision-making, e.g. past and present governance and policy decisions, access to land resources, global markets, and in-migration (Lambin et al., 2003; Lui et al., 2007). As such, care needs to be taken in assuming linear / single pathways explaining LULCC, as land-cover changes are caused by multiple, inter-related factors, with the suite of explanatory forces varying according to specific human-environment conditions at multiple scales of organisation (Lambin et al., 2003).

2. Spatial analysis:

The field of landscape ecology is founded on the relationship between spatial pattern and ecological processes (Turner, 1989; Nagendra et al., 2004), with considerable research focussing on the ecological consequences of fragmentation on natural ecosystems (reviewed in Fahrig, 2003; Fischer and Lindenmayer, 2007; Swift and Hannon, 2010). However, analysing landscape pattern has also provided insight into the human dimensions of LULCC (reviewed in Nagendra et al., 2004). For example, pattern analysis has provided an indication of the growth process within urban sprawl, economic factors driving land demand (e.g. Wickham et al., 2000), migrant colonisation and household development, with changes in household composition affecting land-use strategies (e.g. Moran and Brondizio, 1998; Deadman et al., 2004), as well as the social processes determining patterns of agricultural expansion (Nagendra et al., 2003).

In recent decades, spatial landscape metrics have been successfully utilised as a mechanism to quantify landscape pattern. Their usage has largely focused on the changes in land-cover and land-use, and the consequences for landscape character and functionality (Dale and Kline, 2013; Uuemaa et al., 2013), but they have also been applied to landscape regulatory functions (e.g. fire risk, water quality, flood and erosion control; reviewed in Uuemaa et al., 2013), as well as in predictions of scenic perception and landscape visual character (e.g. Ode et al., 2008; Palmer, 2008).

A significant advantage of landscape metrics is the ease in which they can be applied to categorical map patterns, and the speed at which they can be computed (Uuemaa et al., 2013), a factor influenced by the software available to perform analyses. FRAGSTATS could be considered the most successful of these (McGarigal et al., 2002; McGarigal et al., 2012); a public domain, stand-alone software program, originally released in 1995 (McGarigal and Marks, 1995), and designed to compute a range of landscape metrics measured at multiple scales of landscape analysis.

FRAGSTATS has over 1500 citations in the literature (measured through Google Scholar), but a number of alternative programs are also readily available as stand-alone or as GIS software extensions (e.g. Aue and Ekschmitt, 2010; reviewed in Zargozi et al., 2012).

There is considerable debate in the literature about issues linked to metric redundancy and/or correlation (e.g. Ritters et al., 1995; Cushman et al., 2008; Pend et al., 2010), the sensitivity of metrics to real land-cover changes, with the ecological relevance of landscape metrics often not empirically supported (e.g. Ritters et al., 1995; Li and Wu, 2004; Neel et al., 2004), as well as the effect of scale on metric performance (e.g. Saura and Martinez-Millan, 2001; Simova and Gdulova et al., 2012). Thus, metric selection should be considered carefully, and linked to the specific analysis task, or scale of observation (Zargozi et al., 2012). Metric usage, in the context described in this section, should be considered a *tool* to promote landscape understanding, rather than the primary goal of analysis (Uuemaa et al., 2009).

3. Temporal analysis and analysis of the change process:

Quantifying the process of change requires the measurement of the rate of land-cover change over the analysis period, assessing the progression of change and the spatio-temporal sequences of land-cover change across the landscape over time (Petit et al., 2001).

Given the dynamic nature of human-environmental relationships, time-series analysis provides greater insight into the dynamics of LULCC change compared with snap-shot / point observations (Wackernagel et al., 2004). As such, an important consideration in the assessment of landscape change is that data acquisitions, and thus landscape characterizations, have the temporal frequency necessary to capture change events (Lunetta et al., 2004).

Transition matrices (also ‘cross tabulation matrices’) indicate the changes between land-cover over successive observation points have been used successfully in the literature (reviewed in Aldwaik and Pontius, 2012), and form the fundamental basis in analysis of the change process

(Pontius et al., 2004), helping identify the trajectories of LULCC change over time (Mertens and Lambin, 2000). Typically transition matrices have been used to identify the most important transitions between land-cover classes, i.e. quantifying the temporal sequences of spatial change, and guide research to the processes that may have generated these transitions in the landscape (Pontius et al., 2004). However, Pontius et al. (2004) established that if research focuses only on the largest transitions between land-cover classes, they are likely to ignore systematic exchanges, sequences that are also revealed, but under-utilised, within the transition matrix (Alo and Pointius, 2008).

Evolving from the initial work by Pontius et al. (2004) on transition matrices, Aldwaik and Pontius (2012) developed a methodology, *Intensity Analysis*, which assesses the rate of overall change for the landscape, computed simultaneously for multi-time intervals. This methodology quantifies the dynamics within and between land-cover classes (magnitude, rate and intensity of change), and identifies how landscape transition patterns have altered over time.

This approach has been reviewed in further detail in Chapter 5.

4. Modelling future changes

The literature continues to stress the need to identify future threats to biodiversity, the least of which is because of the 'implementation crisis' (Cowling et al., 2004) that constrains the success of management interventions. For conservation decision-making in particular, identifying management priorities, i.e. the extension of formal protection, scheduling of actions, is of little use if it is only executed years into the future, when threat conditions and biodiversity conservation requirements have altered. As such, effective environmental management must make allowance for any spatial adjustments in land use threats that may possibly occur in the near future. In this regard, predictive land-use modelling is a likely priority.

There are a number of modelling approaches in the literature (reviewed in Franklin, 1995; Guisan and Zimmerman, 2000; Lambin et al., 2000; Veldkamp and Lambin, 2001; Brown et al., 2004), each with benefits and limitations in use (Verburg et al, 2004). Yet in terms of informing policy responses to the potential consequences of future land-cover change (Lambin, 1997), models that are spatially explicit are essential (Lambin, 1997; Irwin and Geoghegan, 2001; Verburg et al., 2002, Brown et al., 2004).

Lambin et al. (2000) have detailed the utility of various spatially explicit modelling approaches on the basis of short- versus long-term predictive potential.
At the broadest level of classification these include:

i) *Stochastic, Empirical, or Statistical* models (e.g. transition probability models, spatial statistical (GIS) models, multivariate models), which, due to their focus on only land-cover transitions observed in the past, have short-term predictive power;

ii) *Process-based, mechanistic models* are based on the interactions between biophysical and socio-economic variables and simulate the evolution of these interactions across the landscape. These models typically have longer-term predictive power;

iii) *Analytical, Economic, Agent-based models* (e.g. optimisation models*)* which identify the underlying causes and scenarios for future changes, i.e. the influence of policy measures on land strategies, and exhibit complex feedback processes, with agent behaviour affecting other agents in the system, as well as the environment (see Brown et al., 2004).

Rouget et al. (2003b) argue that the choice of the most appropriate method is scale-dependant, and determined by the phenomena under consideration, and the data / knowledge that is available. However, even an appropriate model may fail to yield satisfactory results due to poor data, an incorrect resolution of the spatial scale and an inefficient sampling design (Guisan and Zimmermann, 2000). Mapping future land-use pressures is complex, yet lessons from systematic conservation planning in South Africa demonstrate that keeping predictive time frames short (5-10 years), avoiding very complex statistics, and incorporating some expert knowledge into the procedure are ways to ensure that the modelling process is manageable, and will generate good quality data sets (Driver et al., 2003).

1.3 Research aims, objectives and outline of the thesis

The aim of this research is to quantify and provide further understanding into the spatio-temporal patterns of land-cover change in the Kruger to Canyons Biosphere Reserve (K2C), specifically the multi-use landscape mosaic that lies exterior to the western boundary of Kruger National Park. Central to this aim are the implications of the patterns of change to further fragmentation of matrix land-cover and the degradation of existing environmental values, as well as the validity of the current application of the Biosphere Reserve model - not only for this South African example, but also for other sites designated within the Man and the Biosphere Reserve (MAB) Programme internationally.

Three broad objectives result from this:

1.3.1 Objective 1: Critical review of the Man and the Biosphere Programme, and the Biosphere Reserve approach at global scales (addressed in Chapter 2)

- Track the evolution of the MAB Programme from first inception to its current application

- Assess the extent of the Biosphere Reserve theory-reality gap using the status of existing BRs globally, and examine the factors that contribute to the failures and successes in implementation

- Assess the suitability of the Biosphere Reserve designation as an effective conservation approach, exploring the dual conservation-development goals that underlie the guiding principles of successful implementation

1.3.2 Objective 2: Investigate the spatio-temporal dynamics of land-cover change across a multi-use conservation landscape

- Quantify land-cover change in the subregion, providing the regional socio-ecological context for implementation of the Biosphere Reserve concept (addressed in Chapter 3)

- Identify which land-cover classes are the main contributors to observed change dynamics, and how these relate to the landscape utilisation priorities of the Biosphere Reserve model (addressed in Chapters 3 and 4)

- Quantify the spatial dynamics of land-cover change across the subregion, assessing the spatial process of land-cover transformation (addressed in Chapter 4)

- Identify the likely future distribution of land-cover in the subregion (addressed in Chapter 5)

1.3.3 Objective 3: Sustainability considerations of observed land-cover changes for informing management decisions and the long-term effectiveness of the Biosphere Reserve model in the subregion

- Determine the potential consequences of the observed land-cover change trends for the subregion's continued functioning as a UNESCO-MAB 'Biosphere Reserve' (addressed in Chapters 4 and 5)

- Identify a potential / theoretical threshold of future landscape change in K2C that may indicate a signal for scheduling of landscape-scale management intervention (addressed in Chapter 5)

1.4 Structure of the thesis

The thesis is comprised of four content chapters (Chapters 2 – 5) and an introductory and concluding chapter (Chapters 1 and 6).

The content chapters have been written in a format that is suitable for submission to a scientific journal.

Chapter 2 is published in the journal *Biological Reviews* (Coetzer et al., 2013a), Chapter 3 in the *South African Journal of Science* (Coetzer et al., 2010*)*, and Chapter 4 in the journal *Environmental Management* (Coetzer et al., 2013b). The order of chapter presentation in this thesis does not coincide with the time-line of journal acceptance, and Chapter 3 was published prior to Chapter 2 and 4. Chapter 5 will shortly be submitted to an appropriate journal for consideration.

Due to this, there is a degree of repetition in the description of the study site in each chapter, as well as in the description of methodological considerations and motivation for the study.

Chapter 1 describes the rationale for the study, the study aim and objectives as well as appropriate background reading. I have attempted to minimise any overlap in the literature provided in Chapter 1, with that which has been reviewed in the content chapters, and instead direct the reader to the appropriate section. The description of the study site has not been duplicated in this chapter, yet all analyses in the chapters that follow exclude the portion of the Kruger to Canyons Biosphere Reserve (K2C) found within Kruger National Park.

Chapter 2 critically reviews the Biosphere Reserve Model, and provides the wider context for the chapters that follow. Chapter 2 includes some mention of the findings of the subsequent Chapters 3 and 4, as K2C is used as a case-study in the review. While this may appear counter-intuitive to include findings of chapters that follow, given the different journal publication / acceptance dates of each paper, this was unavoidable.

Chapter 3 quantifies the historical land-cover changes observed across the subregion (1993 – 2006), while Chapter 4 is a spatial analysis of important land-cover transformations, in the context of the entire landscape, as well as the Biosphere Reserve's prescribed spatial zonation.

Chapter 5 updates the original time-series, with the additional of 2012 data, and quantifies the changes in the land-cover dynamics across the entire temporal extent (1993 – 2006 – 2012).

Chapter 5 also predicts future land-cover distribution, and identifies potential levels of change that may signal the need for management intervention.

The final chapter, Chapter 6, synthesises the findings from all previous chapters, and discusses the implications of the research for overarching Biosphere Reserve management objectives.

Portions of this research have been included in an additional publication that is not presented in this document. Details of this book chapter are presented below:

Erasmus B.F.N, Coetzer K.L, Mambo J., Archer E.R.M, Fisher J.T., Asner G.P. (2011). *Environmental change in Bushbuckridge.* In: Zietsman H.L. (ed.) Observations on Environmental Change in South Africa, pp. 20-26. Sun Press: Stellenbosch. ISBN: 978-1-920338-24-4.

1.4.1 Author contributions

The paper versions of Chapters 2 – 4 have multiple co-authors. The following list includes a description of the contributions of each author:

- K.L.Coetzer: Primary author, conducted all data extraction and analysis, responsible for paper concepts and the write-up

- E.T.F Witkowski: PhD Co-supervisor, provided guidance with regard to theoretical approach for the paper, commented on various drafts of all papers / chapters

- B.F.N Erasmus: PhD Co-supervisor, provided guidance with regard to theoretical approach for the paper, commented on various drafts of all papers / chapters

- B. Reyers: Commented on initial PhD proposal and commented on the final draft of the paper version of Chapter 4

- A. Bachoo: Provided data pre-processing assistance for initial data sources utilised within Chapter 3

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Chapter 2

Reviewing Biosphere Reserves globally: effective conservation action or bureaucratic label?

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Reviewing Biosphere Reserves globally: effective conservation action or bureaucratic label?

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ABSTRACT

The Biosphere Reserve (BR) model of UNESCO's Man and the Biosphere Programme reflects a shift towards more accountable conservation. Biosphere Reserves attempt to reconcile environmental protection with sustainable development; they explicitly acknowledge humans, and human interests in the conservation landscape while still maintaining the ecological values of existing protected areas. Conceptually, this model is attractive, with 610 sites currently designated globally. Yet the practical reality of implementing dual 'conservation' and 'development' goals is challenging, with few examples successfully conforming to the model's full criteria.

Here, we review the history of Biosphere Reserves from first inception in 1974 to the current *status quo*, and examine the suitability of the designation as an effective conservation model. We track the spatial expansion of Biosphere Reserves globally, assessing the influence of the Statutory Framework of the World Network of Biosphere Reserves and Seville strategy in 1995, when the BR concept refocused its core objectives on sustainable development. We use a comprehensive range of case studies to discuss conformity to the Programme, the social and ecological consequences associated with implementation of the designation, and challenges in aligning conservation and development. Given that the 'Biosphere Reserve' label is a relatively unknown designation in the public arena, this review also provides details on popularising the Biosphere Reserve brand, as well as prospects for further research, currently unexploited, but implicit in the designation.

Key words: biodiversity conservation, Man and the Biosphere Programme (MAB), interdisciplinary science, protected areas, socio-ecological systems, spatial zonation, sustainable development, trade-offs, UNESCO.

CONTENTS

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I. INTRODUCTION: THE BIOSPHERE RESERVE (BR) CONCEPT

The Biosphere Reserve (BR) model of UNESCO's Man and the Biosphere Programme (MAB) represents a shift towards people-centred conservation, explicitly acknowledging humans, and human-interests, in the conservation landscape. Through the establishment of BRs, MAB sets a basis for improving the relationships between people and their environment, a long-term approach that increases people's ability to manage environmental resources sustainably into the future (UNESCO, 1996). Theoretically, where individual BRs contain more than one protected area, they may also fulfil the spatial obligations for more effective protected area system under global-change scenarios, translating the management of isolated 'island' protected areas into regional co-operation, i.e. a regional meta-reserve system that includes matrix areas and enhances connectivity across the BR landscape.

Conceptually the BR model is attractive, yet the practical reality is likely to be challenging, particularly given the history of disappointments of traditional Integrated Conservation and Development Projects (ICDPs; Brandon & Wells, 1992; Neumann, 1997; Wells & McShane, 2004; Blom, Sunderland & Murdiyarso, 2010). Additionally, given that each BR remains under the sovereignty and legislation of the country in which it is found, the BR designation does not guarantee the effective implementation of the concept (Walker & Solecki, 1999). This may be especially true for developing countries where the need for socio-economic development and poverty alleviation is prioritised over nature conservation. Thus, a 'Biosphere Reserve' may remain only a bureaucratic label, with little resemblance to the model envisioned by UNESCO and MAB; the requirements of the designation can be ignored by the State and management objectives of the individual protected areas contained within the BR. As a result, the BR often has a theoretic character (Nolte, 2008), with a considerable gap between the BR concept and the reality worldwide (Price, 2002).

Herein we discuss the history of the Biosphere Reserve concept, and its evolution and expansion across the globe. We assess the extent of the BR concept-reality gap using the status of existing BRs worldwide, and examine the factors that contribute to failures and successes. We examine the suitability of the designation as an effective conservation model, exploring the conservation-development link that is central to the BR concept, and discuss the applicability of the model for developing nations, using the Kruger to Canyons (K2C) BR in South Africa as a case study.

II. BACKGROUND TO THE BIOSPHERE RESERVE CONCEPT

The Biosphere Reserve concept was launched almost four decades ago, in 1974, with the first official international 'Biosphere Reserve' designated two years later. In 1976, some 58 BRs were recognised worldwide (IUCN, 1980). By 2012, this had increased to 610 reserves in 117 countries (Fig. 1; also see Section IV.1 and Fig. 4 for spatial distribution globally).

The Man and the Biosphere Programme (MAB) was launched in 1971 (Dyer & Holland, 1988), after the 1968 'Conference on the Rational Use and Conservation of the Resources of the Biosphere' (Batisse, 1986), as a progression from the International Biological Program (IBP; Di Castro, 1976). While this 1968 conference is referred to as the 'Biosphere Conference' (UNESCO, 1993), there was no reference to 'Biosphere Reserves' (Batisse, 1986), nor, had the notion of 'sustainable development' yet achieved widespread acceptance in the conservation literature (Batisse, 1997). However, this conference paved the way for MAB, emphasising the 'sustainable management of land resources', and prioritising 'the utilization and preservation of genetic resources'. A particular focus was the preservation of a 'representative sample of significant ecosystems, original habitats and remnant populations' (Batisse, 1986, p. 161), which even today is a compliance criterion of BRs (see UNESCO, 1996, p.16).

From the outset, MAB was intended as an interdisciplinary and international approach to resolve ecological and resource management problems (Di Castro, Hadley & Damlamian, 1981; Batisse, 1997), centred around three major themes: (*i*) conservation of genetic resources and biological diversity; (*ii*) international research and monitoring; and (*iii*) ecologically sustainable development (Batisse, 1986; Price, 2002). These themes translated into conservation, logistic and development roles, and BRs were, and continue to be, MAB's instrument to fulfil these roles.

Thus the BR designation was born as the mainstay of the MAB conservation approach (Dyer & Holland, 1988). Implemented through the World Network of Biosphere Reserves (WNBR; Price, 2002), BRs were established as a network of environmentally significant sites, selected for the conservation value of the one or many protected areas they contain, as well as their ability to provide relevant scientific research that would contribute to sustainable development (von Droste, 1987). They were intended as ecological baselines against which the consequences of human-driven modification and management interventions could

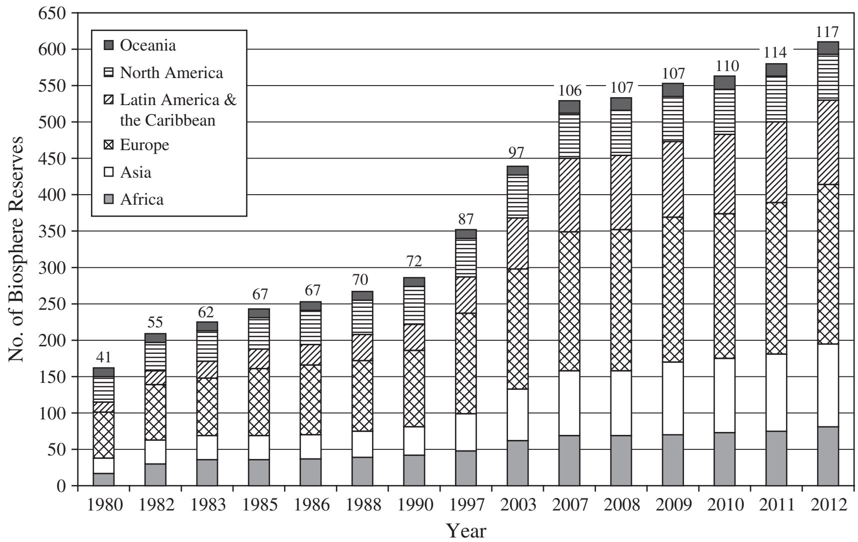

Fig. 1. Biosphere Reserves (BRs) by continent over time (1980–2012) including a country-count. Country counts (above bar) indicate countries as they existed at that date (prior to modern dissolution, secession or renaming. Data extracted from lists compiled by IUCN/UNEP and UNESCO's MAB Programme (1980–2012: (IUCN, 1980, 1982, 1983, 1986, 1988, 1990, 1998, 2003; UNESCO—MAB Secretariat, 2007, 2008, 2009, 2010, 2011, 2012)). Data presented are years for which official protected area lists were available.

be monitored (Di Castro, 1976). The data generated from research within individual BRs would have international applicability and wide-scale relevance (Dyer & Holland, 1988). At inception, BRs were thought to be invaluable in focusing interest and collaborative effort in this intergovernmental scientific endeavour—acting as regional and logistical 'base stations' for any proposed conservation and research plans (Batisse, 1986).

More recently, the BR concept has evolved to represent an interdisciplinary philosophy to address the ecological, social and economic dimensions of biodiversity loss (www.unesco.org). The modern-day BR concept places more emphasis on the relationships between people and biodiversity (Price, 2002), and the protection of socio-ecological systems (SES). People and their activities are considered an integral part of the wider conservation landscape (Brunckhorst, 2001), where environmental protection need not be at the expense of socio-economic development.

Given that strict environmental protection and development are not usually mutually exclusive; BRs have a generalised spatial zonation of acceptable land uses relative to proximity to conservation areas. Different land uses fall into zones of permissible access and enforced controls. A 'core' zone(s) of priority conservation areas and strict formal protection is adjoined by a clearly delineated 'buffer' zone (originally 'inner buffer zone'). In the buffer zone only those activities and land-ownerships compatible with environmental protection can occur (e.g. other conservation designations without strict, statutory protection, ecotourism, recreation). A flexible 'transition' zone (originally 'outer buffer zone') or 'zone of cooperation' (see Price, Park & Bouamrane, 2010) adjoins this area; here, other more intensive land uses are allowed (e.g. human settlements, agricultural areas, traditional resource use) (Batisse, 1986; UNESCO, 1995).

This spatial zonation need not be concentric, and in reality the spatial arrangement of the zones will match existing land-use patterns and development potential in the region. Thus, the 'buffer' zone(s) is one end of the open-ended (and continuous) 'transition' zone (Brunckhorst, 2001), with decreasing intensity of land use and management interventions with proximity to the 'core' areas.

III. DO DESIGNATED BIOSPHERE RESERVES MEASURE UP?

When the first set of BRs was designated, the conservation role was prioritised. Development and logistic roles were overlooked, and links between environmental resource use and development were not addressed (Batisse, 1986). Despite an intervention in 1984 ('*The Action Plan for Biosphere Reserves*'; (Batisse, 1985)), by 1995, there were concerns about a widespread lack of conformity of established sites with the ideals of the BR concept. Only 50% of the 324 existing reserves consisted of a protected core area with additional buffer and/or transition zones, and there was little opportunity for local communities to become active stakeholders in the decision-making process (Price, 2002). As a result, for many sites, BRs were simply a misrepresentation of 'just another' conservation initiative that excluded community involvement; an alternative label for the 'strict protection' typical for legally protected reserves.

Recognising this divergence between the theoretical concept and the *in situ* reality, following the Seville

Erratum: "Given that strict environmental protection and development are often mutually exclusive ..."; [ERROR IN TEXT: 'not usually' corrected to 'often']

Conference in 1995 [which gave rise to two documents that remain at the centre of MAB's position on BRs: '*The Seville Strategy*' and '*The Statutory Framework of the World Network of Biosphere Reserves*' (hereafter 'the Statutory Framework'; UNESCO, 1996)], the periodic review process of existing BRs was initiated (Price, 2002; Price *et al.*, 2010). The period following the Seville conference reflects a shift in the manner in which BRs were defined—a shift in focus from individual protected areas linked to the three zones towards ecosystem/regional conservation with sustainable development as an overarching priority (Ishwaran, Persic & Tri, 2008). The Madrid Action Plan for Biosphere Reserves (2008–2013, MAP) is the current strategic document for existing BRs, building upon the Seville Strategy, with the goal to raise BRs to 'the principal internationally-designated areas dedicated to sustainable development in the 21st century' (UNESCO, 2008, p. 3).

(1) The Seville Strategy and Statutory Framework: conforming to new criteria

The Statutory Framework states that the review procedure should be conducted at 10-year intervals, i.e. on the 10-year anniversary of the designation date or subsequently submitted reviews (Price *et al.*, 2010). It evaluates compliance with the specific criteria (related to size, zonation, ecological characteristics, biodiversity importance, and regional sustainable development prospects; see Statutory Framework, Article 4; UNESCO, 1995) that allow individual BRs to meet the basic conservation, development and logistic roles expected of a 'site of excellence' (UNESCO, 1996, p. 16)' (Price *et al.*, 2010). Failure to fulfil these criteria may eventually lead to a site's UNESCO 'Biosphere Reserve' status being revoked.

Following review, seven countries have removed a total of 12 BRs from the WNBR (Table 1; also refer to Price *et al.*, 2010). All have been voluntary removals by member states, themselves recognising divergence between the status of the BR and the ideals of the BR concept. To date, UNESCO has never forced the withdrawal of a site, but rather recommends specific revisions to meet the criteria of the Statutory Framework. If the BR concerned still fails to comply within a 'reasonable period of time' following these recommendations, it could only then be formally removed from the WNBR (UNESCO, 1995: Article 9).

The Seville conference and its key outputs, the Seville Strategy and the Statutory Framework, represent a point of divergence in the MAB Programme, with a clear division between pre- and post-Seville sites. The majority of existing BRs remain pre-Seville sites (57% in 2010, 54% in 2011, 51% in 2012), with spatial biases in distribution across continents (Fig. 2). The predominance of pre-1995 sites is not unexpected given that the criteria for establishing a BR have become more rigorous over time, but of these pre-Seville sites, only 65% had undertaken a periodic review by 2010, and of these, half did not fulfil the current criteria (UNESCO, 2010*b*). By 2010, over 20% of countries had never addressed the periodic review process or submitted a periodic review report (Price *et al.*, 2010).

However by 2011, the number of sites submitting period reviews had improved, with 28 BRs submitting periodic review reports for the first time (32 submitted in total; UNESCO, 2011*b*). By 2012, a further 33 reviews were submitted, with only nine of these resubmissions by specific BRs, i.e. successive reports addressing recommendations issued by the MAB from previous reviews (UNESCO, 2012). However, the United States, which has more BRs than any other country (47 sites, all designated before 1995), has still not undertaken any official review of its existing reserves (Price *et al.*, 2010; UNESCO, 2010*b*), and few of these [e.g. Southern Appalachians, Champlain-Adirondack (McDonnell, 2005) and the New Jerseys Pinelands BRs (Walker & Solecki, 1999)], appear to have the required spatial zonation.

In this regard, for these sites and a number of others that are in the same situation, in 2011, at the 23rd session of the International Co-ordinating Council of MAB in Germany, members of the Advisory Committee requested that countries that had not submitted periodic reviews be formally withdrawn from the WNBR (UNESCO, 2011*b*). By the 24th session, there was a decision that the MAB-Secretariat would write to all countries concerned, requesting that MAB member states inform on the process of upgrading existing BRs, providing an indicative plan and/or a timeline for the upgrade to Statutory Framework criteria (UNESCO, 2012).The deadline for reply was December 2012.

There is some urgency in ensuring compliance of existing sites to the vision outlined in the Seville Strategy and Statutory Framework, particularly for 'first generation' sites listed during 1976–1984 (see Ishwaran *et al.*, 2008; UNESCO, 2012). The Madrid Action Plan (UNESCO, 2008), is a strategic document in this regard, setting a specific deadline for compliance to the requirements of the 'new' BR concept, i.e. the end of 2013.

(2) The Biosphere Reserve criteria and existing conservation designations: a conflict of interests?

On the basis of the periodic review reports submitted most recently to UNESCO (e.g. UNESCO, 2011*b*, 2012), recommendations made by UNESCO most frequently focus on *inter alia*: enhancing/establishing links between conservation and sustainable development functions, addressing concerns that the conservation function is being undermined by developments in the BR, addressing concerns related to the small spatial extent of a BR and its ability to fulfil function, issues linked to incomplete or ineffectual zonation, insufficient community involvement, and inadequate management plans for the BR.

This is certainly not an exhaustive list of reasons that may result in a site's eventual removal from the WNBR should they not be addressed, nor are they the only reasons that have historically led to a Member State's voluntary withdrawal of a BR from the WNBR. Yet, for those BRs that have been withdrawn from WNBR since the periodic

Table 1. Biosphere Reserves (BRs) removed from the World Network of Biosphere Reserves (WNBR) following review

Biosphere Reserve	Country	Year designated	Year delisted	Reasons for delisting
Northeast Svalbard	Norway	1973	1997	No resident human population within the BR (Price *et al.*, 2010).
St. Kilda	UK	1976	2002	Internationally important site with range of conservation designations, including World Heritage. Spatial restrictions on developing proper zonation, given strict 'Conservation' obligation of the site. Limited community engagement (Scottish Natural Heritage, 2000).
Rùm	UK	1976	2002	No zoning structure present [island managed purely for conservation purposes (Environmental Change Institute (ECI), 1998)]. Scope to introduce three zones severely limited. The majority of the island would require strict, formal protection for conservation purposes (Scottish Natural Heritage, 2000).
Claish Moss	UK	1976	2002	Limited stakeholder involvement; existing core area did not protect an adequate range of habitat types (Scottish Natural Heritage, 2000)
Caerlaverock	UK	1976	2002	Only fulfilled conservation function: small size, no defined buffer zone and limited human population (ECI, 1998). Stakeholder involvement severely limited. The expansion required to meet post-Seville criteria would encroach into priority conservation RAMSAR site where 'multi-use' requirements of zonation unacceptable (Scottish Natural Heritage, 2000).
Taynish	UK	1976	2010	Not possible for the site to meet the Statutory Framework criteria (Price *et al.*, 2010). Limited local population (ECI, 1998) with poor opportunities for stakeholder involvement. Problematic to develop the transition zone (Scottish Natural Heritage, 2000)
Moor House-Upper Teesdale	UK	1976	2012	Site does not meet Statutory Framework criteria UNESCO, 2012. It is comprised of two National Nature Reserves, and forms a broad partnership with the North Pennines Area of Outstanding Natural Beauty, the expansion required of the current core zone was deemed inappropriate (UK—MAB, 2012).
Maritchini Ezera	Bulgaria	1977	2002	Withdrawn by Bulgarian authorities; no review completed before delisting (Price *et al.*, 2010).
South West National Park	Australia	1977	2003	BR consists of only a core area, designated as a National Park and managed for conservation purposes. The protected area's existing National Park and World Heritage designations competed with the BR's 'multi-use' status (Matysek *et al.*, 2006).
Macquarie Island	Australia	1977	2011	No resident human population (Parks and Wildlife Service, 2006)
Bayerischer Wald	Germany	1979	2007	Required extension to meet post-Seville criteria. There was a lack of support from local communities and relevant institutions to be included within the 'new' BR boundaries (Price *et al.*, 2010; UNESCO, 2010*b*)
Lake Torne Area	Sweden	1986	2010	Lacked appropriate zoning (consisted of only a core area of National Parks) with no resident human population in surroundings areas (Thorell *et al.*, 2005).

review process was initiated, these sites have been removed around issues linked to zonation (i.e. only a core area with legal protection), the absence of permanent settlement and stakeholder involvement in the BR, and/or conflicts between conservation and development goals (Table 1). If these are reasons significant enough to result in formal withdrawal from the WNBR, then it appears that many more BRs than those currently listed in Table 1, or marked as 'under revision' by MAB, are also failing to meet the criteria. At least twenty-two sites currently designated by MAB have no permanent resident population, while more do not adhere to the spatial zonation outlined by the BR concept, with non-existent or incomplete zonation (Table 2).

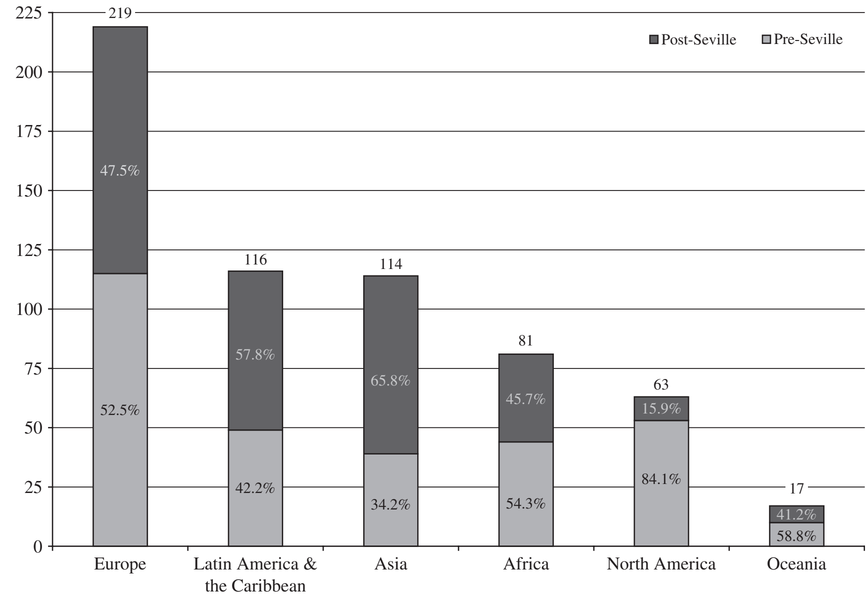

Fig. 2. Pre- and post-Seville differences in the number of Biosphere Reserves (BRs) in 2012 across continents (United Nations macro-geographical regions). Above-bar counts indicate number of BRs by macro-geographical region. Total BRs for 2012 = 610.

A study by Ishwaran (2012) has come to the same conclusion; stating that many existing BRs have been given the 'Biosphere Reserve' designation as simply an additional label to their existing conservation designation or research purpose.

For these BRs (i.e. examples listed in Table 2), park management authorities alone represent the resident human population, and environmental protection, research and tourism and/or nature appreciation is the only 'resource-use' ongoing within the BR's boundary. In these instances, the separate, and more specific, conservation designation that existed prior to the designation of the BR, may better fulfil the management requirements of these reserves than the multi-use landscape requirement of BRs.

The area recognised as a BR may well include multiple designations that appear to serve a degree of the same overarching cause, i.e. environmental protection and long-term sustainability (e.g. Biosphere Reserve, National Park, National Heritage, RAMSAR site, Transboundary Parks). Yet such coincidental designations do not necessarily ensure a cumulative protection effect, especially when strict protection must exist alongside multi-use, sustainable development priorities. Where the zonation structure is lacking, and the BR comprises only a nature reserve or national park with a conservation mandate, it is unlikely that these multiple objectives will coexist.

This was the case for Australia's Macquarie Island BR, which was withdrawn by the Australian authorities at the June/July 2011 meeting of MAB council in Germany, stating the absence of a permanent resident human population as grounds for removal (Table 1). This site had been part of the WNBR since 1977.

The island is listed as critical habitat for two vulnerable albatross species, and inscribed as a World Heritage Site in 1997 (Parks & Wildlife Service, 2006). Macquarie offers invaluable opportunities for scientific research on sub-Antarctic systems (see http://www.antarctica.gov.au/living-and-working/stations/macquarie-island), and given its significance for global geo-conservation and biological diversity, it has severe tourist restrictions in place to ensure minimum human impact on the environment. Tourist visits must be pre-approved, for educational purposes only, and even then, there are spatial and temporal limitations on access to the area (Parks & Wildlife Service, 2006). Given that the MAB approach encourages sustainable, multi-'use' of the landscape (synonymous with some form of resultant land-cover/land-use change), the BR designation seemed incompatible with the level of protection required for this island - particularly given required compliance with the post-Seville criteria, i.e. buffer and transition zones associated with a resident human population. Thus, the conservation of this sensitive environment was likely to be achieved better under its pre-existing, and strictly controlled, World Heritage, Australian Critical Habitat, National Estate and Nature Reserve listings (Parks & Wildlife Service, 2006), rather than one that encourages 'use' of this landscape. The MAB 'Biosphere Reserve' label, in this context, became unnecessary. The former St. Kilda BR, UK, presents a similar scenario (Table 1).

(3) Conforming to the Man and the Biosphere Programme (MAB) as an evolving concept

With the evolution of the BR concept, and the change in focus towards the relationship between conservation and development, it is reasonable that many of these BRs that were established early in the BR concept's history do not conform to contemporary criteria. However, the action

Table 2. Sample of existing Biosphere Reserves (BRs) that at present do not fulfill zonation, resident population, nor economic activities/opportunities criteria post-Seville. Original data obtained from the MAB WNBR directory[a] updated with 2012 withdrawals

Biosphere Reserve (country)	Year listed	Size (hectares)	Other designations	Zonation, with zone sizes (ha) indicated when present	Resident population	Management priorities/ongoing land use
Coram (USA)	1976	3019	National Wilderness	No zonation	None	Research: 'outdoor laboratory'
Three Sisters (USA)	1976	80900	National Wilderness	No zonation	None	Control site for research, recreation
*Circeo (Italy)	1977	8500	State Nature Reserve, Nomination World Heritage, RAMSAR site, Barcelona Convention Specially Protected Site	Core (3 200), buffer (~2800), transition (2500)	~50 inhabitants	Conservation, tourism
*Collemeluccio-Montedimezzo (Italy)	1977	637	European Biogenetic Reserve, State Nature Reserve (IUCN category V)	Core [550 (400 marine)], buffer (241)	None	Conservation, tourism
Iles Zembra et Zembretta (Tunisia)	1977	791	National Park, Specially Protected Area of Mediterranean Interest	Strictly protected (core)	~10 inhabitants	Conservation
Lobau (Austria)	1977	1037	Ramsar Site, Natura 2000, part of Donau-Auen National Park	Undefined	None	Research, education, recreation
Macchabee/Bel Ombre (Mauritius)	1977	3594	National Park	Undefined	None	Tourism, important conservation area, environmental education
North-East Greenland (Denmark)	1977	97200 000	National Park	Undefined	None	Important research site (International Tundra Experiment; www.geog.ubc.ca/itex)
Odzala (Congo)	1977	110000	National Park	Undefined	None	Conservation, tourism. Traditional subsistence agriculture adjacent to park
Mount Chatkal (Uzbekistan)	1978	57360	State Nature Reserve	Core (45160), buffer (12200)	None	Conservation, research
Torres del Paine (Chile)	1978	184414	National Park	Core only	None	Managed as a national park; tourism, research
Laguna San Rafael (Chile)	1979	1742 000	National Park	Undefined	None	Tourism, research
*Pietrosul Mare (Romania)	1979	46399	Scientific Reserve within Rodna Mountains National Park	Undefined until 2010; renaming and extension: core (8200), buffer (11800), transition (24000) (UNESCO, 2011*b*)	None	Conservation

Table 2. Continued

Biosphere Reserve (country)	Year listed	Size (hectares)	Other designations	Zonation, with zone sizes (ha) indicated when present	Resident population	Management priorities/ongoing land use
Gorge of Samaria (Greece)	1981	4850	National Park	Undefined	None	Restricted tourism
Mount Olympus (Greece)	1981	4000	National Park	Core and buffer only (zone sizes not specified)	None	Managed as a National Park; tourism (core); grazing, hunting, wood extraction (buffer)
Kronotskiy (Russian Fed.)	1984	1142134	State Reserve, World Heritage Site	Undefined (under revision)	None	Research, limited tourism
*Waddensea of Lower Saxony (Germany)	1992	240000	National Park, World Heritage Site	Core (130000), buffer (108000) [extended transition zone (2000) proposed]	One researcher	Conservation, tourism

*indicates intent to adjust BR to meet criteria in near future (MAB-Italy, 2011; UNESCO, 2011*b*).

[a]http://www.unesco.org/new/en/natural-sciences/environment/ecological-sciences/biosphere-reserves/world-network-wnbr/wnbr/

required to reflect the requirements of post-Seville BRs may be beyond the capacities of BR managing authorities, or may not receive the appropriate stakeholder buy-in from surrounding communities.

In the case of Bayerischer Wald BR, Germany, which was removed from the WNBR in 2007 (Table 1), stakeholders in the area adjacent to the BR strongly opposed their inclusion into the redesigned BR. Enlargement of the site was opposed by the local community, and the BR was withdrawn from the WNBR as a result (Price *et al.*, 2010; UNESCO, 2010*b*). While the conservation-development model is attractive, and may offer local residents previously unavailable opportunities to benefit from conservation in the BR, the restrictions imposed on land uses within the BR boundaries may be unattractive compared to land uses found outside, making inclusion into the 'new' Reserve unappealing (see Section IV.2).

UNESCO (2010*b*) has also found that the costs associated with the review process are a limiting factor in the success of the transformation process, with one Australian signatory from Tasmania's Department of Primary Industries, Parks, Water & Environment, stating that the effort spent on developing reports for MAB could be better utilised on much needed conservation projects within the core zone's National Park (Price *et al.*, 2010; UNESCO, 2010*b*). This particular BR, Australia's South West National Park, was removed from the WNBR in 2003 (Table 1).

Price *et al.* (2010) quantified these costs into the thousands (US dollars). While the effort spent on preparing periodic review reports was not necessarily comparable across countries, costs ranged from $2200 (Canada, where time was donated by national experts and only travel costs were measured) to $43,000 (France).

However, regardless of the difficulties associated with review and the subsequent restructuring of non-conforming sites, the review process is currently the only mechanism that exists to assess the WNBR as individual sites and as the cumulative network. It is the only mechanism to gauge MAB's progress in reconciling biodiversity conservation and sustainable development (Price, 2002; Price *et al.*, 2010).

Some sites have been transformed effectively following review, and there are a number of success stories for transformation internationally (see UNESCO, 2010*b*, 2011*b*, 2012). In particular, Braunton Burrows BR, UK, now enlarged and renamed 'North Devon BR', after revision, became the UK's first 'new style' BR (Macleod & Price, 2012). It is considered to epitomize the model BR by many (Ishwaran *et al.*, 2008).

(4) UK-MAB: a success story for the review process

Following the success of the North Devon revision, UK-MAB invested further in ensuring compliance of their remaining BRs, all established in 1976, setting the precedent for the revision process in the other MAB member states.

In England, this process resulted in the withdrawal of one of its three BRs, Moor House-Upper Teesdale BR (Table 1). In Wales, the Dyfi Valley BR, the only BR designated here, was enlarged and renamed (now *Biosffer Dyfi Biosphere)*; its boundaries extended beyond the National Nature Reserve that had previously accounted for the entire Biosphere region (www.unesco.org). With this revision, Biosffer Dyfi effectively became the UK's second 'new style' BR (Macleod & Price, 2012).

In Scotland (home to the remainder of the UK's BRs, as well as five BRs now withdrawn from the WNBR, Table 1), two of the original BRs (Cairnsmore of Fleet, and Merrick Kells-Silver Flowe) previously not fulfilling the Statutory Framework criteria individually, were included in a single, larger 'Galloway and Southern Ayrshire BR'. This extensive consultative process began in 2006 (see Wallace, 2011 for details), and following its renomination to UNESCO in 2011, this site was successfully added to the WNBR in July 2012 (http://www.unesco.org).

The Beinn Eighe BR, which failed the revised criteria due to insufficient human intervention and poor opportunities for sustainable development (Hambrey Consulting, 2009), would potentially conform if local communities living outside the BR allowed for expansion of reserve boundaries, i.e. beyond only the Beinn Eighe National Nature Reserve. The Wester Ross Alliance (http://www.wester-ross-alliance.co.uk) initiated dialogue with other regional stakeholders, with a proposal to create a large 'Wester Ross BR' with the current Beinn Eighe site as part of its core (Macleod & Price, 2012). The enlarged site would qualify under post-Seville criteria with little change to existing institutional structures and land management (Hambrey Consulting, 2009; Macleod & Price, 2012), making it an excellent candidate for reinstatement. However, the challenge for this particular BR, still in the proposal stage, is in acquiring commitment for the required financial investment/resources; the current economic climate may favour expenditure in the established conservation areas rather than newly designated sites (see Macleod & Price, 2012).

Given that a significant proportion of conservation initiatives in the UK are community led with voluntary organisations playing an important role in environmental protection [conservation authorities in the UK own only approximately 3% of their National Parks, with public bodies, charities and private alliances accounting for the remainder (Selman, 2009)], there is a history of sympathetic stakeholder involvement associated with conservation. This is likely to favour the discussions around the revision of the remaining BRs, and ultimately the sustainability and successful implementation of MAB in the UK long term.

IV. CONSERVATION AND SUSTAINABLE DEVELOPMENT

(1) Political buy-in: the attraction for developing country economies

Ultimately, the BRs fall under the jurisdiction of individual member countries, regardless that it is an international designation and a global collaboration. The individual protected areas within the BR may be afforded the protection of the state, but the region defined as the BR frequently does not have the same supporting legislation to ensure its persistence. Thus for the BR model to be successful long term, it requires political buy-in at the level of state/provincial or national government, and if popular political benefit is seen to be absent, it is unlikely that these governments will continue to support the ideals of the BR concept (J.D. Brown, 2002*a*).

Prior to the era of 'new conservation' (K. Brown, 2002*b*, 2003), protected areas, and their associated fortress conservation ethic, typically evoked negative social responses, with a narrative of displacement and/or expulsion, exclusion and resource restriction accompanying their creation (West, Igoe & Brockingham, 2006, for review; Hartter & Goldman, 2010). Parks were seen to be perpetuating the poverty cycle, further marginalising the already marginalised [e.g. the San and Bakgalagadi of Botswana's Central Kalahari Game Reserve (Hitchcock, 2002)], and establishing an enduring attitude of resentment towards conservation agencies (Brockington & Igoe, 2006; Rangarajan & Shahabuddin, 2006; West *et al.*, 2006; Hartter & Goldman, 2010). However, the widespread political acknowledgement for more accountable conservation decisions makes the people-centred conservation model more politically congruent, for both developed and developing country governments alike. Similarly, the awareness for 'sustainability' is rising and therewith so too is political interest around the concept.

The attraction of the BR concept is that it offers a mechanism for the traditionally protectionist conservation land use to 'work for the poor'. In a climate where social equality and economic upliftment is prioritised, it becomes politically dangerous to favour environmental protection over socio-economic development.

Since inception, the BR concept has been especially favourably received by developing country governments [approximating 'Emerging' and 'Developing economies', as categorised by the International Monetary Fund (2012) World Economic Outlook], with the average rate of 'buy-in' through the listing of new BRs each year exceeding that of developed countries ('Advanced economies'; IMF, 2012) pre- and post-1995 (Fig. 3). The difference in the total cumulative listings in each period is significant between developed and developing countries ($X^2 = 19.02$; d.f = 1; $P < 0.00001$; Fig. 3), becoming noticeably more pronounced in developing countries over time, i.e. when the Seville Strategy refocused the objectives of the BRs on sustainable development, and the rate of new listings increased accordingly. These differences may indicate the influence of different sets of stakeholder groups as instrumental in the decision-making process around BR proposal and eventual establishment, i.e. conservationists and scientists in developing countries, while citizen groups, local land-owners, social organisations and local councillors are frequently more prevalent in developed countries. Recognising that a BR's establishment may be motivated by different desires, values and/or expectations will be useful in future BR-stakeholder relations

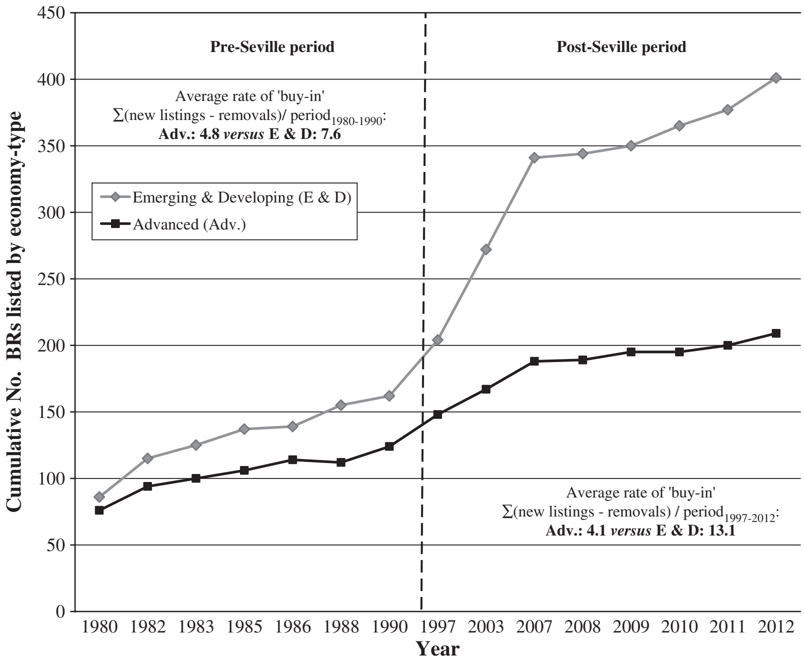

Fig. 3. Cumulative Biosphere Reserve (BR) listing pre- and post-Seville by economy-type. Data presented are years for which official protected area lists were avaliable (IUCN/UNEP and UNESCO's MAB Programme: 1980–2012). Data points in the trend line were calculated as the number of BR by economy type over time (a cumulative total of pre- and post-Seville, accounting for new listings and removals from the WNBR for each year); the average rate of 'buy-in' was calculated for periods: 1980–1990 and 1997–2012, where $t_0 = 1980$ for the pre-Seville, and $t_0 = 1997$ for the post-Seville period.

(see Bouamrane, 2007 for case studies across different socio-ecological contexts; see also Sections IV.2 and IV.3).

As of 2012, 65.7% of the listed BRs have been established in emerging and developing countries (Fig. 4). The remaining 34% of BRs are located in advanced-economy countries, with the majority (61.2%) of this portion established pre-Seville.

(2) Biosphere Reserves and links to Integrated Conservation and Development Projects

The linking of development with conservation is not unique to the BR concept. While the Seville Strategy may have reorganised the ideals of the MAB so that 'people-centred conservation and sustainable development' became a priority criterion, Integrated Conservation and Development Projects (ICDPs) had already begun to attempt this a decade previously.

The first ICDPs were launched in 1985 by the Wildlands and Human Needs Program (WWF) as a mechanism to integrate economic development with sustainable natural resource management, targeted specifically at the 'rural poor' (Hughes & Flintan, 2001). Today's working definition of these original ICDPs may have evolved somewhat, but contemporary versions (i.e. the 'new conservation') remain primarily biodiversity conservation projects that combine social development (and enhanced livelihood options) with environmental protection (Hughes & Flintan, 2001).

The history of ICDPs has been one of elusive success, with the literature indicating that they have failed their promise, i.e. 'conservation by distraction' (Ferraro & Simpson, 2000). Confusion over objectives, vague assumptions, naïve expectations and a failure to acknowledge trade-off between conservation and development priorities have led to significant criticism of the concept by social scientists and ecologists alike (reviewed in Hughes & Flintan, 2001; Robinson & Redford, 2004; McShane *et al.*, 2011). These criticisms include: (*i*) that projects act as population 'attractors', fuelling migration to project areas, and hence (*ii*) result in more rapid land-cover change around conservation areas; (*iii*) resource entitlements/restrictions as a result of projects shift resource exploitation to new areas; (*iv*) that projects are frequently linked to external donor funding cycles which raises issues for project sustainability; (*v*) projects result in biased and disproportionate benefits across communities; (*vi*) there is limited participation with decisions made by 'external' parties with little understanding of social and ecological context; and (*vii*) the mismatch between timeframes for economic and biodiversity outcomes prevent the successful integration of objectives.

As a result, resurgent protectionists have begun to advocate more strongly for the dissolution of coupled protected area-development initiatives, and the return to strict authoritarian enforcement in conservation areas (Sandker *et al.*, 2009; see Wilshusen *et al.*, 2002 for discussion of the 'resurgent protectionist' argument). There is also increasing support for the replacement of the traditional notion of ICDPs by 'landscape or ecosystem approaches' (Sayer *et al.*, 2006; Sayer, 2009; Axelsson *et al.*, 2011). Landscape approaches,

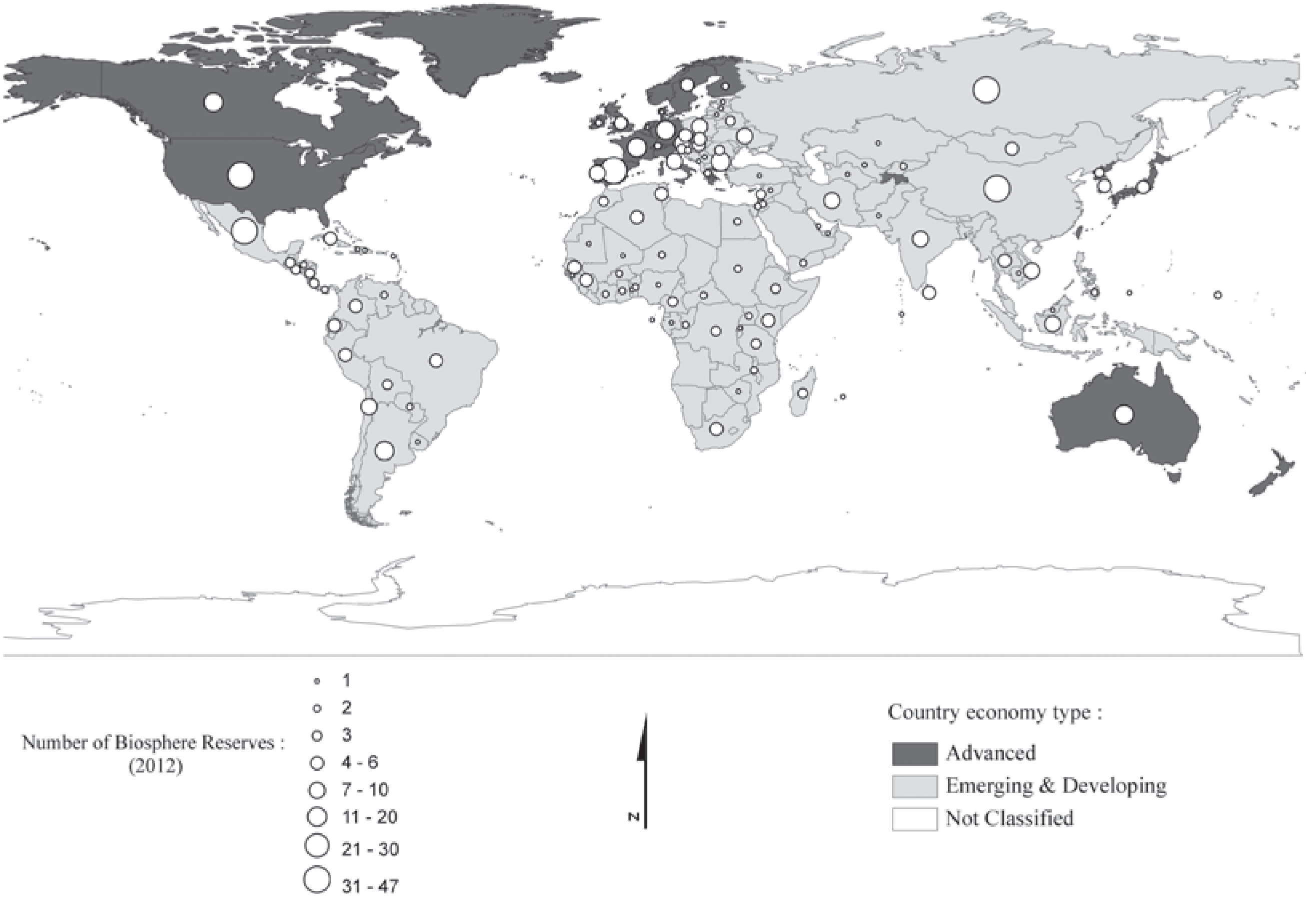

Fig. 4. Relative distribution of 2012 Biosphere Reserves (BRs) by country economy-type. The August 2011 and July 2012 additions included seven countries that had previously not been represented in the World Network of Biosphere Reserves (WNBR): St. Kitts & Nevis, Togo, the Maldives, Lithuania (2011) and Haiti, Kazakhstan, Sao Tome & Principe (2012). All of these are listed by the International Monetary Fund (IMF) as 'Emerging & Developing' economies. 'Economy-type' is defined as in the IMF's 2012 World Economic Outlook (WEO) classification system (International Monetary Fund, 2012; http://www.imf.org).

given their larger scale and explicit appreciation of a diverse landscape mosaic, are thought appropriate for resolving the trade-offs between economic, social and conservation objectives in complex socio-ecological systems. However, a landscape approach for reconciling development with conservation may present its own operational challenges.

As with ICDPs, there is little empirical evidence that a landscape approach is more effective than ICDPs in aligning environment-development objectives (Sayer *et al.*, 2006). Given that it is still largely untested and undocumented, and that it shows convergence with the ideals of the original ICDP approach, there is the potential that a landscape approach, by adding complexity to an already complex aspiration (e.g. aligning conservation and development for the benefit of both) may suffer the same limitations of the traditional ICDP.

In this regard, the BR designation may be especially useful. The development of the ecosystem approach was influenced by the systematic approach of UNESCO-MAB's implementation principles and designation requirements (Axelsson *et al.*, 2011). Given BRs prerequisite of larger size, prescription for co-management and multiple stakeholders in the designation of a site, opportunities for continuous review and interdisciplinary research, and potential for exploitation as 'natural experiments' at a global scale (see Section VII), BRs can provide the framework for action learning around win-win solutions, and ultimately contribute to sustainability in many scenarios.

Thus, the ICDP definition (and future evolution thereof) demonstrates substantial convergence with MAB's current aims and objectives. BRs provide an indirect linkage between livelihoods and conservation through substitute economic activities, provided by the BR authorities in return for compliance with the spatial restrictions on resource use (Salafsky & Wollenberg, 2000). Such development initiatives are as diverse as they are abundant, and examples are included in Table 3.

Many of Table 3's projects remain in their infancy, i.e. Cape West Coast, or have only been implemented as 'pilot projects' thus far, i.e. Kruger to Canyons, Yayu, Lake Vanern BRs, but these economic interventions may help secure confidence in the BR model, and increase

Table 3. Economic development initiatives implemented within selected Biosphere Reserves (BRs)

Biosphere Reserve	Economic development and welfare upliftment initiative
Giam-Siak Kecil—Bukit Batu BR (Indonesia)	Establishment of acacia plantations as a community development initiative (UNESCO, 2010*a*) and to target illegal logging of natural forest; pilot program to initiate bio-village [Asia Pulp & Paper (APP), 2011].
Yayu BR (Ethiopia)	Sustainable agriculture initiatives targeted at coffee production linking local coffee production to organic and free-trade markets (through training, and assistance with coffee value additions, i.e. harvesting, processing, certification and marketing procedures). Agriculture initiatives aimed at honey and fruit production are in the planning and fundraising stages. Renewable energy initiatives (from coffee-waste, small-scale hydro- and solar power) will drive further local development (Deprez, 2011*a*, *b*, *c*)
Yancheng BR (China)	Programs to expand cultivable land area through the construction of aquaculture ponds, reed lands and salt pans in the tidal areas on the eastern side of the BR (Ma *et al.*, 1998). While this expansion guaranteed habitat transformation, the activities remained tolerable for the vast resident waterbird population while food and water remained in adequate supply. Protecting the bird species population is the BR's main objective.
Bia BR (Ghana)	Local villages have been allocated harvesting licenses for African giant snail (*Acantina acatina*) found in abundance in the BR. Income obtained from this is partially utilised for conservation purposes, with the remainder returned to the villages for use in community-based projects, such as for water pumps and improvements to schools (Schaaf, 2009).
Nanda Devi BR (India)	Afforestation projects and the provision of mechanical soil conservation measures, solar power, improved beehive and spinning devices at subsidised prices. Government support for establishing vermicompost and the use of biofertilisers—in anticipation of increased demands for organic crops grown in the BR (Saxena *et al.*, 2010).
Sierra Gorda BR (Mexico)	Establishment of one of the first (social) carbon projects developed by the Grupo Ecologico Sierra Gorda operating within the carbon market's 'very voluntary' arena (i.e. no third-party certification). Offers institutions the option of offsetting carbon emissions by giving financial compensation to farm-holders who pledge reforestation and environmental conservation (Deprez, 2011*d*). Project has direct benefits (education, training, payments for ecosystem services and conservation of private lands, water storage works) for >200 communities in the BR, with conservation benefits of >13 000 hectares returned to natural forested state (http://www.carbonneutralplanet.org/basket.pdf). Other sustainable agriculture initiatives successfully established value-added products: e.g. *Sierra Country Products* (intensive management and sale of livestock products), aquaculture products, *Pure Life Foods* (organic fruits and vegetables), *Green Gold Certified Oregano* and *Honey of Tancoyol* (Deprez, 2011*e*).
Entlebuch BR (Switzerland)	Development of the BR-linked label, '*Echt Entlebuch*' (genuine Entlebuch), identifying high-quality products (e.g. cheese, ham) manufactured within the BR. The Swiss National Tourist Office has awarded three of these products the prestigious ' NaturPur' label. Local restaurateurs (>300) bear the 'Gastropartner' label awarded by the BR; this label identifies local establishments partnering, and utilising Entlebuch Biosphere products (UNESCO Biosphere Entlebuch, 2007; Schaaf, 2009).
Lake Vanern (Sweden)	Projects to link recreational fishing more closely with *in situ* fish sales and more efficient, more holistic processing methods. Currently recreational fishing extracts a comparable volume of fish to commercial fishing from the lake. The pilot project aims to enhance sustainability of the lake's fisheries use by *inter alia* linking food to the BR region, allowing local use of ecosystem services, and improving the lake's management practices (UNESCO, 2011*a*; www.euromab2011.se).
Kruger to Canyons BR (K2C) (South Africa)	Established natural-resource-use projects centred on environmental access rights (www.kruger2canyons.org/projects.html). These projects include a successful *Bio-cultural Protocol* project implemented through traditional health practitioners which outlines the traditional practitioners' role in the community and the challenges they face. Bio-cultural protocols assist in protecting traditional knowledge and traditionally used resources, and help practitioners acquire access to areas that are unavailable for harvesting. This project is closely linked to a community-based carbon-trading project (*Voluntary Carbon Offset Programme*), where carbon is sequestered through the planting and growing of important medicinal plant resources. This helps to address resource availability issues due to over-harvesting or lack of access. The *Environmental Monitor Project*, implemented under National Government's Extended Public Works Programme employs previously unemployed community members from within K2C; specifically to assist with the monitoring function of the BR. K2C also conducted a feasibility study for a hydro project (*Blyde Hydro Project*) as a direct result of collaboration with the Rhön BR in Germany. The project proposes the development of a hydro station on an already established dam wall. However, due to the potential financial rewards and political challenges, this project was taken over by the Provincial authority, with K2C committed to the final decision. The Kruger National Park (KNP) has also implemented its own community-targeted development projects in the BR: a pepperbark *Warburgia salutaris* (IUCN: Endangered) harvesting and propagation (Scheepers *et al.*, 2011), a thatch harvesting (Zambatis, 2011), and a mopane worm harvesting project (Swemmer *et al.*, 2011), allowing approved access to the resources in the National Park.

Table 3. Continued

Biosphere Reserve	Economic development and welfare upliftment initiative
Cape West Coast BR (South Africa)	Education outreach and scholarship programmes. These include youth schools environmental education, scholarships to study at the South African Wildlife College and bursaries for tertiary education. The BR also has begun negotiation of a major industrial corridor through the subregion, and assisted local government with geographic information system GIS projects (du Toit, 2011).

support for community inclusion into the BR. Some projects, for example the original agricultural expansions in Yancheng BR, the initiatives in Entlebuch BR and the carbon project in Mexico's Sierra Gorda BR, have already had successful outcomes (Table 3; Hughes & Flintan, 2001; http://www.carbonneutralplanet.org/basket.pdf), with these development initiatives having positive impacts on poverty while environmental integrity was unaffected or even slightly enhanced.

However, BRs fall within the theoretical arena of 'win-win' conservation and development solutions; true win-win outcomes are rare, with trade-offs between competing objectives likely (reviewed by McShane *et al.*, 2011). Where conservation initiatives inflate community expectations, without being explicit about potential, and possibly disproportionate, livelihood losses and restrictions on resource use, the discord between community members and conservation agencies is likely to be substantial. Wells & McShane (2004) refer to this as a cycle of 'optimism and disenchantment', which, if unresolved, will have long-lasting consequences for future conservation initiatives in the area.

In some instances the potential benefits of these projects are viewed as inferior to the livelihood losses incurred due to spatial restrictions on use, e.g. Nanda Devi (Maikhuri *et al.*, 2001; Saxena *et al.*, 2010) and Bayerischer Wald BRs (Table 1; Price *et al.*, 2010; UNESCO, 2010*b*). This is particularly true when benefit-sharing is viewed, rightly or wrongly, as inequitable across stakeholders, i.e. biased towards a select few in the community (e.g. individuals qualifying for participation in pilot projects). Many of those who may benefit from conservation-linked development opportunities, e.g. ecotourism initiatives, may not be the ones who are adversely affected by the change in resource access rights, while those who suffer the costs may not be sufficiently reimbursed for their sacrifices (Fuentes-Quezada, Sekhran & Kunte-Pant, 2000; Fu *et al.*, 2004).

For example, local farmers in the Wolong BR, South Western China bore the conservation costs of restricted land use, with their traditional resource use impaired by the spatial regulations of the reserve (Fu *et al.*, 2004). They received no tangible economic rewards from the eco-tourism initiatives implemented within the BR, nor were able to afford the hydro-powered electricity offered as an alternative source of energy to the now restricted fuelwood harvests. For this portion of the community, the establishment of the BR further undermined their already fragile socio-economic circumstances, offering no opportunities for improvement of livelihood options.

When the Maya Biosphere Reserve, Guatemala, was established in 1990, the initial eviction and subsequent resettlement of the original residents from the National Park to government-established agricultural properties *('fincas')* was generally well accepted. This resettlement mainly involved the *Comunidades Población en Resistencia* (CPR), a faction opposing the Guatemalan government during the 36 year civil war, who had sought refuge in the National Park. However, the relationship between Maya BR and new colonists moving into the reserve from other regions of Guatemala continues to be fraught with difficulty. The ongoing displacement of the new colonists and residents from the core zones, and limitations on access elsewhere in the 'multi-use' zones, has led to repeated illegal re-colonisation of the core zones, and ecological sabotage, such as setting fire to the BR to make conservation pointless (McNab & Ramos, 2007). Further compounding the lack of regard for the Maya BR designation is the government's inability to police and evict the wealthy landowners from the BR, which has precipitated a wider disregard for the Maya BR's rules and regulations in general. As a result, stakeholders of the BR have suggested *inter alia*, that the BR be de-proclaimed/eliminated, or that the core area's restrictions on access be reassessed, with the core zones re-designated as forest concessions (McNab & Ramos, 2007).

In the Changbai Mountain BR, China, until new legislation and a more effective management organisation prohibited such activities in 2006, reserve authorities supplemented their limited salaries by offering entrepreneurial activities, issuing collection contracts to private individuals for forest resources (e.g. pine nuts). These contracts generated significant resentment towards both the contractors and reserve authorities, and provided local residents with the motivation to continue with their own exploitation of the forest, regardless of the restrictions in place (see Yuan, Dai & Wang, 2008, for details).

The examples listed here are not representative of all the stakeholder-management relations found across the WNBR. It is certainly not the case for those BRs that have undergone review and subsequent transformation successfully; for example, the UK examples in Section III.4, the examples Reed & Egunyu (2013) discuss from Canada, or in the Kristianstad Vattenrike Biosphere Reserve, Sweden, where extensive research in co-management has been undertaken to improve the 'fit' of social institutions and the natural environment (see Olsson *et al.*, 2007). Yet these examples illustrate the fragile relationship between BR managers, local government and resident communities, and the problems that may arise if managers treat BR residents as a homogenous entity, or enforce decisions without appropriate participation.

Results from ICDPs have shown that projects that fail to address equity issues (i.e. gender issues, economic discrimination) and social organisation, particularly local decision-making hierarchies (i.e. family hierarchies, traditional authorities, such as chiefs and councils) have had reduced success, especially in terms of project longevity (reviewed in Hughes & Flintan, 2001). These same cautions must apply to BR management. In the case of the Maya BR example, the modern BR may need to re-evaluate access rights to generate support around the designation (McNab & Ramos, 2007) which raises challenges for what appears to be already weak or discriminatory governance.

(3) Local stakeholder buy-in

Thus, BRs that fail to address participation effectively will undermine long-term sustainability, as shown by Stoll-Kleeman & Welp (2008). In a survey of 213 BRs, increased participation resulted in increased social acceptance of the BR which, in turn, resulted in more effective conservation. BR managers cannot neglect appropriate and unbiased community engagement, regardless of the complexities that may be involved with the participatory process, e.g. the large numbers of stakeholders involved in decision-making and the opportunities for conflict in the management process. In this regard, the BR concept has provided a systematic approach to achieving effective stakeholder-manager dialogue, using detailed examples of the consultation and co-decision process in selected BRs (reviewed in Bouamrane, 2007).

However, in this era of people-centred conservation, BRs may also find invaluable insights from tools that are traditionally linked to the marketing and product development arena given that conservation has moved beyond biogeographical problems, to include sociological issues central to success. In this context, programs like Future Search (http://www.futuresearch.net/), Open Space (http://www.openspaceworld.com/users_guide.htm), and World Café (http://www.theworldcafe.com/) have merit for participatory management (reviewed in Stoll-Kleeman & Welp, 2008) with the Open Space method already used by UNESCO Venice with some success. Other methods that may offer insight to the relationship between communities and their BR, include focus groups [see Stoll-Kleeman & Welp (2008) for explanation], key informant interviews, and participatory geographic information system (GIS) approaches [e.g. for discourse around resource allocation in BRs; see Kwaku Kyem (2004) for review of participatory GIS applications].

V. CAUTIONS FOR BIOSPHERE RESERVES: CHALLENGES IN ALIGNING CONSERVATION AND DEVELOPMENT

BRs may offer prospects for socio-economic development as an incentive to the BR stakeholders, yet development delivers its own challenges for these conservation areas. Encouraging development in sensitive environments may have negative consequences not only for ecological integrity and conservation values but also for economic activities that depend on an appreciation of an 'unspoiled' environment and the notion of 'wild nature', i.e. for recreational purposes.

(1) Consequences for ecological integrity

In Wolong BR the presence of a road system allowed residents to access biodiversity resources a distance away from their villages (Fu *et al.*, 2004), exploiting areas that were previously inaccessible to residents, and fuelling more rapid land-cover change across the landscape. Once intact habitat is lost from the landscape on a large-scale, e.g. due to development initiatives, the cycle is frequently self-propagating involving land-cover change trajectories of increasing transformation with further development (Brady *et al.*, 2009). While the notion of development itself is not problematic, unregulated and indiscriminate development agendas will compromise the BR long term, particularly at the level of spatial zonation (see Section VI.2).

In Yancheng BR initial development programs expanding cultivable land area did not negatively affect biodiversity (Ma *et al.*, 1998; Table 3). However, this success led to further development projects, effectively intensifying development activities in the BR, yet assuming the same negligible environmental impacts as the first project. Conservationists foresaw local species extinctions as a result (i.e. river deer *Hydropotes inermis*), and the forced migration of the waterbird population as the wetland ecosystem was irrevocably changed (Ma *et al.*, 1998). The new development permissions allowing for over-development of the BR (Ma *et al.*, 2009) resulted in economic rewards for the local BR human population. Yet, these new permissions had serious consequences for the environmental integrity of remnant wetland habitat, and the priority BR species (Table 3), the endangered red-crowned crane (*Grus japonensis*), now only survives in artificial wetlands created in the core of the BR (Ma *et al.*, 2009).

With dual conservation and development priorities, the challenge is for BR authorities to identify change trajectories and react appropriately, pre-empting unfavourable transformation before it results in irreversible environmental degradation. The reality of effectively aligning conservation and development is likely to be a dynamic balancing act between conservation *versus* development objectives over time. Sensible choices need to consider the (changing) socio-ecological context of the landscape, and involve repeated reassessment of potential impacts of development projects, i.e. a pragmatic and adaptive management approach that responds to defined thresholds of acceptable/unfavourable land-cover transformation. Remotely sensed data will be a useful tool in this regard (see Section VI.2).

(2) Consequences for economic circumstances and the Biosphere Reserve 'brand'

Where the development activities planned for a BR detract from the notion of a pristine environment, i.e. a common expectation of a 'National Park' or 'Nature Reserve', the consequences for tourist visitation may be severe, especially when these activities impact the formally protected areas inside the BR. BR authorities may need to adopt precautionary measures to ensure that economic development projects do not diminish the conservation values (tourism, biodiversity protection) of existing protected areas, particularly when the BR label as an attractor for tourism is frequently overshadowed by the protection status of the protected areas within the BR.

For example, in the Lake Torne BR in Sweden, before it was withdrawn from the WNBR in 2010 (Table 1), tourists were unaware of the BR, and were attracted to the region by the presence of Abisko National Park (Reinius & Fredman, 2007). In Slovakia, Hungary and the Czech Republic, the BR label is largely ignored by tourists, and even by local conservation authorities, compared with other conservation designations in the area (Nolte, 2004). In Australia, excluding the recent additions to the national BR network, BRs have been traditionally perceived (by tourists, as well as by federal decision-makers) to be areas that do not uphold environmental protection due to their emphasis on development. They have been seen as competing with existing national conservation programmes [e.g. National Heritage Trust 'Care' Program (1997–2008, now: 'Caring for our Country')], thought to be an unnecessary addition to existing conservation policies, and challenged for limited funding allocated for National Parks (see Matysek, Stratford & Kriwoken, 2006, for specific details).

However, this is not the case for all BRs, and in the Entlebuch BR, Switzerland, for example, the reverse is true. Broad-based image campaigns have raised awareness for the BR and its 'back-to-nature' style tourism. Targeted tourism marketing concentrated on the subregion has also been utilised extensively. Entlebuch BR is well accepted as the paradigm of Swiss regional parks, with the Swiss media frequently (almost daily) reporting on the BR and its various economic activities, specifically the high-quality food products bearing the Entlebuch identifier, and the BR's 'Gastropartner' concept (UNESCO Biosphere Entlebuch, 2007; see Table 3).

Similarly, in the Bañados del Este BR, Uruguay, an eco-labelling initiative identifies certain products produced in-line with the BR's sustainable approach to resource-use, and the direct involvement of local residents in product development and marketing. Besides the promotion of locally sourced products, the label, '*productos Banados del Este*', also highlights the territory as unique in the region, i.e. a 'Biosphere Reserve' (Bouamrane, 2007). In the Grosses Walsertal Biosphere Park, Austria, BR branding links local agricultural products to the Biosphere Park (e.g. timber, agricultural brands: 'Walserstolz', 'Bergtee', 'die kostliche Kiste') with the local hospitality industry awarded the label '*Partner Company of the Biosphere Reserve*'. Although the direct tourist value from this branding remained insufficient, it elicited a strong positive response from the partners involved; effectively enhancing co-operation between local businesses across the BR (Lange, 2011). As such, it is incorrect to restrict the opportunities in BR branding to only marketing specifically for increasing tourist awareness. Rather, BR-linked product (and service) labelling may have an important role to play in creating and strengthening the regional identity of local communities located within BRs (see Lange, 2011).

Thus better marketing of the BR label may adjust people's perception of the designation. The BR designation is a relatively new label compared to those of 'National Park' and the other well-known protected areas within the BR region. There will be a lag period before BRs have the same effect on tourist visitation as these other, better known conservation areas (Reinius & Fredman, 2007). Such marketing initiatives need not be expensive; with social media, i.e. *Facebook*, *Twitter*, *LinkedIn*, *YouTube*, BR authorities can easily reach, and popularise their reserves with international audiences. The viral nature of the online community will ensure self-propagation of the BR campaign. The newly launched *BiosphereSmart* website initiative may be instrumental in this regard (http://biospheresmart.org/). The interactive portal is a global observatory created to share experiences and best practices among BRs. It is based on the idea of maximising (new) information technologies to strengthen partnerships across the WNBR, as well as empowering communities by providing access to information and enhanced decision-making capacity, i.e. with citizens acting as beneficiaries *and* actors in this BR information-sharing arena.

Clever use of the MAB mandate in these campaigns, i.e. social justice ideals, development initiatives that preserve ecological and cultural integrity, aggressively linked to contemporary issues in the traditional media, may target the social conscience of potential visitors, i.e. concepts of environmental responsibility. This should generate discourse amongst like-minded internet users, and mainstream the BR concept more rapidly.

VI. BIOSPHERE RESERVES AND PRACTICAL IMPLEMENTATION: THE CASE OF THE KRUGER TO CANYONS BIOSPHERE RESERVE (K2C), SOUTH AFRICA

Given the potential benefits of the BR concept for developing countries outlined in previous sections, generating broader public support and goodwill towards this 'new' conservation approach may need to become a priority. The history of conservation decisions has been fraught with ethical injustices (reviewed in West *et al.*, 2006; Redford & Fearn, 2007), with many of these historical actions still having negative effects today, e.g. protected area establishment tied to longer term poverty, cultural changes and subsistence losses (Rangarajan & Shahabuddin, 2006; West *et al.*, 2006). BRs need to ensure that they do not follow the same trends—given that displacement from the core zones or limitations on access rights may accompany designation.

In the South African conservation context, there is a long history of discriminatory displacement and eviction associated with protected area establishment, which has given rise, in some cases, to enduring attitudes of resentment. In fact, the Kruger National Park (KNP), South Africa's flagship protected area for biodiversity conservation, when it was first established was seen 'as a means of providing more effective control over neighbouring Africans, as well as those who remained in the park' (Carruthers, 1995, p. 65). The establishment of the park prioritised white South African interests over those of other races, with the description 'National' and 'Kruger' equivalent with 'white' culture and heritage at this time (Carruthers, 1995).

Thus, for a large portion of the South African population, protected areas, KNP in particular, are seen as symbols of 'apartheid repression' rather than viewed with national pride. There have even been calls for KNP's de-proclamation, stating that its 20 000 km^2 of untransformed African savanna would be better utilised as agricultural land for the impoverished communities who live immediately outside its boundaries (Carruthers, 1995).

However, recent systematic biodiversity assessments have also flagged these areas immediately west of KNP as requiring urgent additional conservation in response to land-use pressures (e.g. Driver *et al.*, 2005; Ferrar & Lotter, 2007). Given the historical inequalities in the subregion, and current socio-economic circumstances, the further exclusion of local communities to enhance levels of biodiversity protection will be met with strong opposition. Thus, for the South African situation, the BR model may be especially appropriate.

The Kruger to Canyons Biosphere Reserve (K2C, http://www.kruger2canyons.org) of South Africa includes the iconic KNP, and the aforementioned impoverished communities located adjacent to it. The apartheid era displacements have had long-term repercussions for the social fabric of this sub-region, and the BR designation here has been applied with the hope of redressing the inequalities in socio-economic circumstances across the landscape. Should this model be effective in this regard, it would have broader relevance for other parts of South Africa, or other countries where similar circumstances prevail, e.g. China (e.g. Yuan *et al.*, 2008), India etc., where large portions of the population still rely on direct harvesting for daily livelihood demands, and/or live in close proximity to sensitive conservation areas. This section details the difficulties of establishing a BR in an already well-established cultural landscape, and suggests a management strategy for the future that will better align the BR model with the situational context of the South African political, social, economic and conservation environment.

(1) Background to the subregion

K2C is located in the north-eastern part of South Africa (Fig. 5A, B) and is one of the largest BRs in the world, at approximately 2.6 million ha in extent. Established in 2001, K2C includes a diverse array of economic and conservation land-use types, and has a resident human population of >1.5 million people, most of whom reside in the transition zone in rural communities.

Conservation accounts for more than half of the available land area in the BR, while in the unprotected remainder, economically important (nationally and regionally) commercial agriculture, mining (e.g. Palabora mine, the largest open cast copper mine in southern Africa) and forestry sectors persist. The settlement areas continue to expand. In places population densities in these settlements exceed 300 people/km^2 (Pollard, Shackleton & Carruthers, 2003), despite the semi-arid environment.

The expansive rural settlement areas in K2C are well-established artefacts of the former apartheid era. The forced relocations of black South Africans to the rural 'homelands' of Lebowa, Gazankulu and KaNgwane (encompassed in the present-day Bushbuckridge and Maruleng administerial districts in K2C; Fig. 5A), and the associated separate development policy of the apartheid regime have had long-term repercussions for the socio-economic circumstances of the communities here.

High population densities, entrenched poverty, and poor economic opportunities have ensured a well-established relationship with the natural resource-base in communal rangeland areas associated with each village to support household livelihoods (Shackleton & Shackleton, 2004; Twine, 2005). Rangeland usage includes traditional agriculture, grazing of domestic stock, fuel- and construction-wood harvesting, and the collection of food items, thatching grass and medicinal plants.

While non-destructive use of the environment should not compromise the ecological integrity of the BR in general, the recent influx of foreign refugees into the region (Polzer, 2007), and the weakening control of the traditional authorities over resource management (Twine, 2005), may compromise the sustainability of this resource extraction long term. Already, recent studies indicate that current levels of timber resource extraction in the BR are unsustainable (Fisher *et al.*, 2012; Matsika, Erasmus & Twine, 2013; Wessels *et al.*, 2013).

(2) The designation of K2C and the situational context

For K2C's resident population, the listing of the site with MAB provides access to previously unavailable economic and livelihood opportunities in the future (see Table 3 for specific examples of projects that have been piloted or are planned for K2C). In terms of future developments proposals, K2C management authorities have established a zone-specific, conceptual framework that guides further development initiatives (http://www.kruger2canyons.org/index.html). These appear to be sufficiently broad to be incorporated into

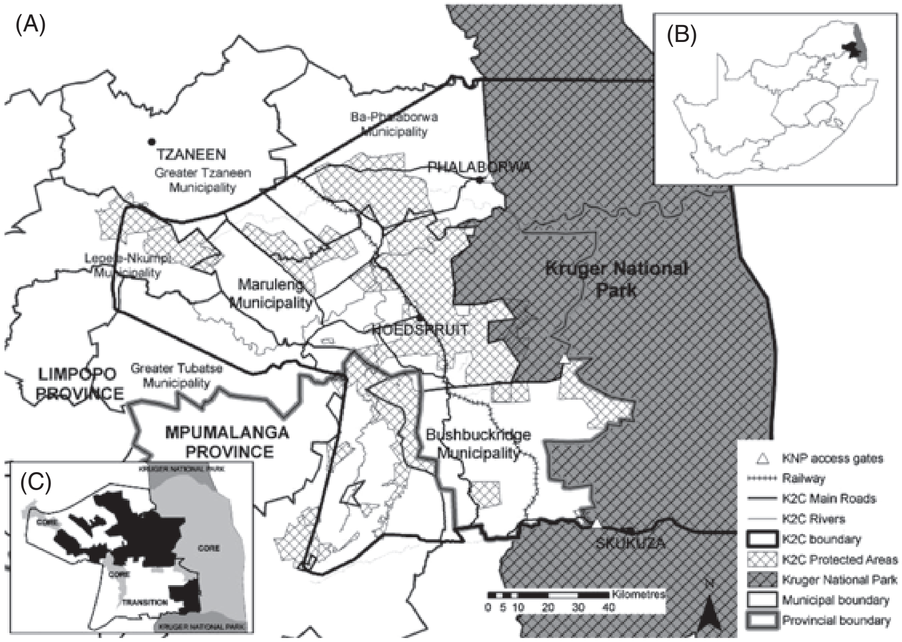

Fig. 5. (A) Regional location of the Kruger to Canyons Biosphere Reserve (K2C), including protected areas, municipal districts, main roads and rivers. (B) Map of South Africa showing provincial boundaries. (C) Spatial arrangement of core, buffer and transition zones in K2C.

national policy around land-use planning. Their practical alignment, however, is likely to be more challenging.

The zonation of this BR is neither the stylised concentric rings of model BRs, nor is it the spatial arrangement of core: buffer: transition zone envisioned by the BR concept (Fig. 5C). In K2C, with reference to KNP and the south-eastern portion of the BR in particular, densely populated settlement areas lie immediately adjacent to the National Park, with negligible buffering between the two land-use types.

Admittedly, settlements were established prior to the listing of K2C, which limits the delineation of the subsequent zonation. Yet, this negligible spatial buffering (note: not 'buffer zone' as in the formal BR zonation), between the National Park, and a growing settlement expanse (with an associated increase in unregulated resource extraction), may eventually compromise the conservation values of KNP, and other protected areas in the BR.

In K2C, between 1993 and 2006, settlement areas increased by 39%, with expansion tracking transport routes (Coetzer *et al.*, 2010). Given the relationship between the local communities and their communal rangeland areas, as settlements enlarged, the footprint of resource extraction increased, with consequences for the extent of intact habitat nearby. Furthermore, the land-cover change occurring in the transition zone appeared to have impacts that extended beyond this zone, encroaching into the protected areas in the buffer zone of central K2C (K.L. Coetzer, E.T.F. Witkowski, B.F.N. Erasmus & B. Reyers, in preparation).

This scenario is not restricted to K2C, and in other developing countries, land-use change in the transition zone has also threatened the longevity of conservation initiatives in the core zone (e.g. Ma *et al.*, 2009). Where solutions were offered in order to address the resultant environmental degradation that occurred as a result of the development process, as in the case of Yancheng BR, they focused on specifically redefining the nature, extent and location of all three zones, given that significant changes had occurred in the transition zone since the BR's first listing (Ma *et al.*, 2009). K2C may find value in considering a similar revisionary approach.

While K2C may have produced guidelines for future land-use in the BR, the available land area for development and expansion is a severe limiting factor, not only for the local settlement communities but also the other land-use stakeholders of the subregion. Limitations on 'free space', together with an impoverished and growing population and their more intensive usage of the local environment, will undermine any theoretical zonation and restrictions on 'use' in the BR. For the expanding settlement communities, the immediate fulfillment of daily living demands may take precedence over any longer-term benefits from the planned development initiatives, with consequences for the land-cover of the BR. This raises questions around the validity of the BR designation, as it has been currently been applied, for K2C's already well-established land-use matrix, particularly around issues of sustainability.

However, the land-cover change presented here has occurred regardless of the subregion's listing as a 'BR', rather than as a result, i.e. due to BR-linked sustainable development objectives fuelling land-use change—given that the site was only established in 2001. K2C has also made noteworthy advancements around stakeholder alignment for resilient socio-economic development (Coetzee & Biggs, 2012). This was an extensive consultation process that set the baseline for a sustainable mixed(land)-use landscape mosaic in the future, and K2C been formally recognized by UNESCO in this regard.

However, it may still be useful to re-evaluate the appropriateness of K2C's zonation in the future, with the BR management exploiting adaptive management techniques which respond to changes in land-use in the BR over time. As such, historical, and future, land-cover change trends will be instrumental in this decision-making process, and should be a priority for this, and other BRs. The feasibility of such an approach for K2C and other developing countries is uncertain, given the limitations on financial and human capital (as opposed to natural capital; Brunckhorst, 2001), but the rewards are likely to be large. A dynamic management strategy may ensure the longevity of the BR into the future, which is likely to have important consequences for the subregion as a whole.

VII. BIOSPHERE RESERVES AS LEARNING SITES FOR INTERDISCIPLINARY RESEARCH

BRs are an ambitious designation when compared to traditional protected areas, and are essentially, when applied effectively, broader socio-ecological systems: cultural landscapes (*sensu* Farina, 2000) that require both social and ecological management strategies aligned for biodiversity protection and environmental sustainability. However, the implicit socio-ecological designation of the BR model raises the additional challenge of practically aligning these two 'cultures of conservation' (Rangarajan & Shahabuddin, 2006). In this regard, biologists/ecologists and social scientists are typically found on separate ends of the same (conservation) continuum, and will focus on different components of the bigger (BR) picture, i.e. people and nature. Research and management priorities of each will be disparate; disconnected by disciplinary preferences (Rangarajan & Shahabuddin, 2006).

Interdisciplinary research has received considerable interest in the conservation literature over time, but even recent publications emphasise the need to integrate the sciences more effectively (e.g. Balmford & Cowling, 2006; Meinke *et al.*, 2006; Margles *et al.*, 2010; Driscoll *et al.*, 2012). While conservationists certainly recognise the importance of an inclusionary approach that incorporates the social dimensions of conservation problems, the perception is that the gap between biological and social/economic/political sciences remains considerable, even with increasing awareness by both 'sides' (Fox *et al.*, 2006). There are few instances where practitioners and researchers have successfully integrated disciplinary approaches (with the possible exception of the field of ecological economics), due to lack of communication between parties (Fox *et al.*, 2006; Margles *et al.*, 2010). It is this gap that undermines effective biodiversity protection as, in general, contemporary conservation has become 'not about biology, but rather people and the choices they make' (Balmford & Cowling, 2006, p. 692); a complex problem that is also affected by development pressures and the prevailing political climate.

Advocates for mainstreaming interdisciplinary approaches more urgently stress the need for more structured opportunities for interdisciplinary collaboration, opportunities that allow collaboration from the project start-up, rather than at the end, i.e. as an ethical after-thought for better marketing with the public and political decision-makers (Fox *et al.*, 2006). MAB's BR model may be the ideal arena in which to do this.

When the BR concept was initiated, it was intended as a pioneer of intergovernmental interdisciplinary research and collaboration (Ishwaran, 2012), and while there may be a delay in the effective realisation of this role at the level of individual BRs, the theoretical framework already exists, i.e. in the form of today's WNBR and associated thematic activities. Herein, the Madrid Action Plan (UNESCO, 2008) specifically recognises the potential for the WNBR to function as 'learning sites for sustainable development'; emphasising BR's role in promoting the exchange of ideas and experiences from relevant site-based research across the global network (UNESCO, 2008). UNESCO's GLOCHAMORE project ('Global Change in Mountain Regions'; see Björnsen Gurung, 2006), and subsequent project, GLOCHAMOST ('Global and Climate Change in Mountain Sites'), are excellent examples of global collaboration and interdisciplinary research that has been undertaken within existing BRs (www.unesco.org). Since GLOCHAMORE was first initiated in 2003, more than 300 scientists from a wide range of scientific disciplines and some 25 BRs participated in the research, which focused on global change in mountain ecosystems and its socio-economic impacts on the livelihoods of mountain people (Lange, 2011; Schaaf, 2011).

The BR conceptual framework is well-established, and the underlying criteria of the concept already compel a relationship between social and environmental aspects in the conservation landscape. Where BR successes have been limited (see Sections IV and V), experience has shown that it is typically a failure on behalf of BR authorities properly to consider different stakeholders in decision-making and project design. Thus, explicitly incorporating a systematic, interdisciplinary approach from the initial stages of BR nomination, as well as more structured practical implementation guidelines, will serve both goals: ensuring the success of individual BR sites long term, and fulfilling the broader research need.

Thus all working BRs have the potential to deliver a solution to the disciplinary divide at the scale of individual sites: an adaptive management approach to develop 'prototype' programs for interdisciplinary (and ultimately, transdisciplinary) research and implementation (see Kleiman *et al.*, 2000 for details of a 'prototype program'; Biggs & Rogers, 2003 for an effective adaptive management approach specifically for conservation and resource management).

These prototype programs and their design principles, where successful, could be expanded for use within other BRs.

Such a structured program approach would address the concerns that the research completed in individual BRs is not being effectively up-scaled for international benefit; that individual BRs are not developing scientific approaches appropriate for other sites, and that the co-ordination of 'learning' is as yet unrealised (see Ishwaran, 2012, for details), regardless that the Madrid Action Plan (2008–2013) has set specific targets for collaborative, site-based, policy-relevant research and 'experience sharing' by 2013. Systematic interdisciplinary research programmes should also increase buy-in for the BR model in general, particularly at the level of acquiring donor funding. This could be crucial to the success for developing country BRs, lacking the financial and human capital for management but with abundant natural capital for conservation.

BRs are conservation designations that foster sustainability through their implementation, and given that they contain formal protected areas at their core, they are not immune to the challenges of environment-development reconciliations (Reed & Egunyu, 2013). However, while BRs are clearly aligned with the general ideals of 'sustainable development', it may be that the BR use of the term 'development' as a core objective of the concept (see Section II), may be intentionally or unintentionally overemphasizing the economic dimension of sustainable development, to the detriment of the social and environmental dimensions. Sustainable development is not merely economic development, but is anchored on the nexus of economic, environmental and social pillars of change. Thus the traditional expectation of 'development' as a specific function of a BR may also be deflecting from the designation's current role as 'learning sites' under the Madrid Action Plan (UNESCO, 2008) or 'learning laboratories' (Ishwaran *et al.*, 2008; Nguyen, Bosch & Maani, 2009). Contemporary BRs are to serve as learning sites for sustainable development: developing models and demonstrating approaches for global, national and local site-based sustainability.

The difference between the two expectations may be subtle (i.e. sustainable development as a core objective *versus* 'learning sites' for sustainable development), but in the contemporary context, the emphasis is on BRs to advance research and learning around sustainable development. They are to facilitate sustainability on the ground, not only in specific sites but also for up-scaling and international application elsewhere. This perception of 'learning sites' encourages experimentation and implies the opportunity for critical reflection on success and failure, an evaluation necessary for effective adaptive management (Schultz & Lundholm, 2010; Reed & Egunyu, 2013). Few existing BRs have the structures in place to fulfil their 'learning site' potential (see Schultz & Lundholm, 2010, for details). However, provided that research results are well documented and communicated, progress will be made towards this eventual goal, e.g. disseminated and shared *via* the *BiosphereSmart* website. The periodic review process may be an existing structure that may also be instrumental in this regard, presenting research results or project outcomes for the benefit of other sites (see Reed & Egunyu, 2013).

Holling (2001) indicates that the term 'sustainable development' is an amalgamation of the meanings implied in 'sustainability' and 'development'; that it is a logical partnership, with the goal of fostering adaptive capacity and creating opportunities for social, ecological and economic understanding. It has however, up until now, been somewhat slow to deliver on its 'promise' internationally. The notion of effective sustainable development goes beyond only what can be typically achieved through isolated integrated conservation-development projects or other conservation-development relations coupled with specific protected areas. Thus, a learning approach that is a global and interdisciplinary collaboration, undertaken across a range of socio-ecological or established natural experiments (in the form of the WNBR) may be the priority, not only for conservation, but for our understanding of complex adaptive systems in general.

VIII. CONCLUSIONS

(1) The backlog of first-generation Biosphere Reserves that do not meet the new requirements of the designation post-Seville remains large; regardless that the number of new sites listed each year continues to increase rapidly. This raises questions whether the current quality control mechanism enshrined in the Statutory Framework is working. It suggests the need for a more rigorous approach in applying, and reviewing conformity to the Biosphere Reserve model.

(2) Biosphere Reserves incorporate conservation areas recognised under other formal conservation designations. However, overlapping designation does not always ensure cumulative protection effort. Where sensitive environments exist, the 'development' and 'resource-use' requirements of the Biosphere Reserve model may not favour these existing conservation areas, and will require continuous re-evaluation.

(3) The Biosphere Reserve concept has been especially well received by developing countries, with the notion of dual conservation and sustainable development objectives attractive for encouraging socio-economic development in conservation landscapes. However, the literature has shown that similar approaches have not been especially effective in previous conservation efforts, i.e. ICDPs, with few compelling examples of success. Yet the literature also stresses the need for more human-centred conservation approaches, and the Biosphere Reserve model, when implemented effectively, is especially valuable in this regard. Already there are successful conservation-development projects existing in a number of listed sites.

(4) A more comparative integration of the 'lessons learnt' from existing Biosphere Reserves may improve implementation success of the model. This review has provided a number of case studies emphasizing ecological

and economic cautions in applying the designation. While a 'one size fits all' approach to implementation is unrealistic, a documented adaptive learning approach considering the successes and challenges of other reserves will be beneficial to both newly listed sites and other, older sites struggling with compliance to the Statutory Framework criteria.

(5) There are a number of excellent opportunities present in Biosphere Reserves, not only for ecosystem conservation and global-change scenarios given their typically larger sizes, but also for structured interdisciplinary research. This literature continues to emphasise the need for better integration amongst the disciplines, with few examples of this occurring successfully. The implicit 'social' and 'ecological' nature of working Biosphere Reserves suggests the potential, if harnessed appropriately, to function as 'prototype programs' for interdisciplinary collaboration. The global partnership, the GLOCHAMORE project has already been successful in this regard.

(6) It may be that continuing to emphasise 'development' as a core function of Biosphere Reserves is deflecting from the importance of the new roles of Biosphere Reserves under the Madrid Action Plan, i.e. as 'learning sites' for sustainable development. While the two objectives are clearly closely aligned, in the contemporary expectation the emphasis is on research and continuous action learning for sustainable development. Sustainable development has been slow in its global delivery. Re-assessing (*i*) the nature of research undertaken in Biosphere Reserves (e.g. as interdisciplinary from the project design phase), and (*ii*) utilizing the WNBR more effectively as established natural experiments for sustainability (as prescribed in the Madrid Action Plan), may be the way to improve the success of sustainable development initiatives worldwide.

(7) Biosphere Reserves are not intended to replace existing conservation designations/actions, but rather to enhance them: to improve the relationship between the environment, society and economic development in the landscapes in which they have been housed. This review has illustrated the potential of the Biosphere Reserve approach. This review has also indicated a number of challenges in applying the Biosphere Reserve concept internationally, with implementation lagging in many examples. However, with the more rigorous approach to evaluating sites since 2011/2012 (see Section III.3), it is likely that those Biosphere Reserves that do not function as the concept intends, i.e. a 'Biosphere Reserve' in label alone rather than in practice, will be withdrawn from the WNBRs in the near future, improving the successes of MAB globally.

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Chapter 3

Land-cover change in the Kruger to Canyons Biosphere Reserve (1993 – 2006): A first step towards creating a conservation plan for the subregion

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Land-cover change in the Kruger to Canyons Biosphere Reserve (1993–2006): A first step towards creating a conservation plan for the subregion

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ABSTRACT

This paper is a first step towards a conservation plan for the Kruger to Canyons Biosphere Reserve (K2C) on the South African Central Lowveld, quantifying the historical land-cover trends (1993–2006). During the analysis period, 36% of the biosphere reserve (BR) underwent land-cover change. Settlement areas increased by 39.7%, mainly in rural areas, becoming denser, particularly along roadways. Human-Impacted Vegetation increased by 6.8% and Intact Vegetation declined by 7.3%, predominantly around settlement areas, which is testament to the interdependency between rural communities and the local environment. However, settlement expansion exceeded the rate of rangeland growth; in the long term, this may raise questions for sustainable resource extraction. Similarly, the block losses of intact vegetation are of concern; issues of fragmentation arise, with knock-on effects for ecosystem functioning. In the economic sector, agriculture increased by 51.9%, while forestry and mining declined by 7.1% and 6.3%, respectively. The future of these three sectors may also have significant repercussions for land-cover change in the BR. The identification of historical drivers, along with the chance that existing trends may continue, will have important implications for biodiversity protection in this landscape. Applied within a conservation-planning framework, these land-cover data, together with economic and biodiversity data, will help reconcile the spatial requirements of socio-economic development with those of conservation.

INTRODUCTION

Land-cover transformation poses one of the greatest threats to global biodiversity[1] and, in light of extensive land-cover change, protected areas (PAs) are often viewed as the 'Noah's Ark' for conservation. However, there has been growing awareness that formal PA systems are failing to protect global biodiversity effectively[2] and additional areas are required to supplement PAs and connect reserve networks. Off-reserve conservation and matrix management has become an imperative.[3] In order to supplement the size of PAs and link PA systems, conservation initiatives have had to look beyond reserve fences, to the actual management of the surrounding landscape to enhance PA function. Yet, given that land-cover transformation may continue unchecked in the unprotected matrix,[4] land-use in this surrounding landscape may not necessarily support conservation objectives.

Many of the current PAs in South Africa that are, in theory, ensuring the persistence of the country's biodiversity, were established for a different purpose (i.e. as scenic tourist destinations), or were either unsuitable for agriculture or located in residual areas under previous governments.[5] The fact that many of the existing PAs were not explicitly created to ensure biodiversity preservation or the long-term maintenance of ecosystem functionality,[5,6] especially under global change scenarios, suggests that the presently limited conservation funds are not being best employed to meet the full suite of national biodiversity objectives. However, action has begun to address the shortcomings of the national PA system (e.g. *The national protected area expansion strategy*[7]), but competing land-uses and development proposals curtail the scope of such conservation initiatives.

Landscape transformation is inevitable with ongoing population growth and socio-economic development, especially as this development has been prioritised by the National Government's electoral mandate[8] until 2014. Given the current Government's emphasis on socio-economic development, conservation has had to evolve in order to be politically acceptable. Conservation decisions cannot occur in isolation any longer; future conservation initiatives must explicitly acknowledge proposed socio-economic development agendas. The challenge now is how to rectify the failings of the current PA system in order to maximise biodiversity protection, whilst addressing the unavoidable land-cover modification that is associated with economic growth. It is here that systematic conservation planning[9] offers significant opportunities.

The central premise of conservation planning is to make informed decisions about the limitations of current PA systems and direct additional conservation action to ensure *enduring* biodiversity protection. Not only do conservation planners need to be aware of how biodiversity features are distributed, but they also require spatially explicit data on current biodiversity threats (i.e. conservation-hostile land-cover and land-uses), as well as data on the rate of land-cover transformation.[10] Biodiversity conservation is more likely to endure if conservation initiatives consider the spatial requirements of other land-use sectors, avoiding, where possible, those areas that will experience a high probability of conversion in the future. Even within land-cover classes, the capacity and attitude of stakeholders (while not assessed in this paper) are crucial to the success of conservation initiatives; enduring conservation goes beyond simply establishing a biophysical template.

This paper is the first step in a much larger conservation planning exercise for the Kruger to Canyons Biosphere Reserve (K2C; http://www.krugertocanyons.com) in the Central Lowveld area of South Africa. Although the area boasts significant spatial investment in conservation, the subregion, particularly the western part of the biosphere reserve (BR) that is aligned with the north-eastern escarpment bioregion, still remains a priority area for conservation action.[11] Furthermore, the area is impacted by extensive human settlement and economic activity and certain areas, particularly the former homelands of the apartheid regime, have been targeted as priorities for future socio-economic development (i.e. so-called 'Presidential poverty nodes').[12] Thus the choice of land-use options in the K2C is a highly contested and, often, an emotive issue; any conservation initiative that failed to include land-cover and land-use trends would be ineffective. Acknowledging the history of land-use in the BR will help to identify transformation probabilities between land-cover classes and thereby pre-empt the possibility of spatial conflict between conservation and other land-uses in the future.

In this paper, we quantify the land-cover changes that may threaten the BR's biodiversity and address the historical land-cover in the subregion through the creation of status quo land-cover maps. The landscape and the landscape trends are described in-depth and presented as the net land-cover change (1993–2006) of the subregion. We also quantify the spatial nature of the land-cover change in the BR, highlighting areas of dramatic change and suggesting the direction future research and conservation planning decisions should take.

METHODS

Study area

The K2C covers approximately 2.6 million hectares and traverses the Limpopo and Mpumalanga Provinces (Figure 1). It includes the central Kruger National Park (KNP) and a suite of privately owned PAs. In total, just under half of the BR is dedicated to formal biodiversity protection, while agriculture, plantation forestry, mining and settlements dominate in the remaining unprotected land.

The climate is subtropical, with hot, humid conditions and summer rainfall (500 mm – 700 mm per year). Winters are mild and generally frost-free.[13] The K2C includes three biomes: Savannah (northern, western and central areas of the BR), Grassland (the southern limb and isolated patches in the west) and Forest (small isolated patches across the higher elevation areas along the escarpment in the west of the K2C). It encompasses four bioregions: the Lowveld and Central Bushveld predominate (Figure 1), but areas of Mesic Highveld Grassland and the Mopane Bioregion are also present.[14] The diverse vegetation and habitat types have ensured remarkable conservation and tourism opportunities and, owing to the extensive formal PAs, the subregion is an important ecotourism destination.

The K2C was registered with the United Nations Educational, Scientific and Cultural Organization's (UNESCO) Man and the Biosphere Programme in September 2001. It is the largest BR in South Africa and the third largest in the world. As with other BRs, the K2C is divided into three distinct zones: a core zone, a buffer zone and a transition zone.[15] The core zone (approximately 900 000 ha) of formal protected areas is dedicated to the strict protection of biodiversity, while the buffer zone (approximately 480 000 ha) adjoining the core areas, only allows for activities that are compatible with conservation objectives. Beyond this lies the transition zone (approximately 1.1 million ha), wherein land-users must sustainably manage and utilise the area to maintain ongoing ecological functionality. It is within this transition zone that the subregion's mostly rural and rurban (having both urban

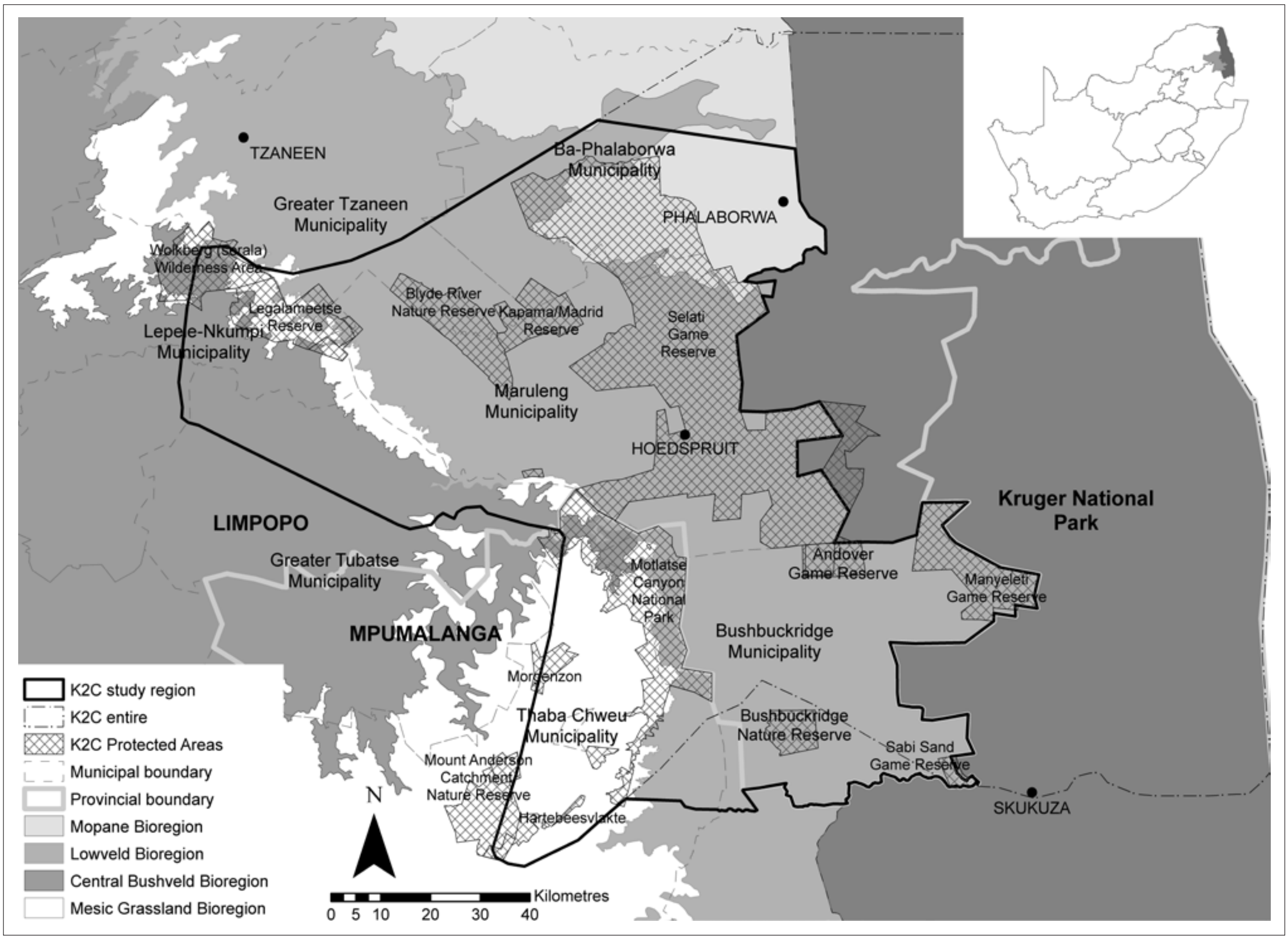


FIGURE 1
Regional location of the Kruger to Canyons Biosphere Reserve (K2C), including protected areas, vegetation bioregions and municipal districts

TABLE 1
Hierarchical description of land-cover classes used in the land-cover classification procedure

Consolidated land-cover class	Land-cover subclass	Description
Exposed Ground and Rock	Exposed Ground and Rock	• Areas of exposed sand, soil, rock
		• Excludes opencast mines and quarries and fallow agricultural fields
Water	Waterbodies and Rivers	• Open water areas, both artificial and natural areas, and dams and rivers
Intact Natural Vegetation	Intact Woodland	• Intact indigenous plant communities suffering limited / negligible anthropogenic modification
		• Moderately open savannah e.g. 'Granite Lowveld' vegetation type[14]
		• Dominated by tall to medium-sized trees with a distinct herbaceous layer
		• Tree species: *Acacia nigrescens*, *A. nilotica, A. excuvialis, Sclerocarya birrea* subsp. *caffra, Combretum apiculatum, C. imberbe, C. zeyheri, Dichrostachys cinerea, Peltaphorum africanum*
		• Herbaceous species: *Eragrostis* spp., *Aristeda* spp., *Digitaria erianthra*, *Panicum maximum*
	Intact Thicket and Bushland	• Moderate to dense woodland with a well-developed shrub layer and distinct herbaceous layer, for example, 'Poung Dolomite- and Ohrigstad-Mountain Bushveld' vegetation type[14]
		• Includes bush encroached areas
		• Tree species: *A. caffra, A. excuvialis, A. eriloloba, A. tortilis*, *Boscia* spp., *C. apiculatum, C. molle, C. hereroense, Dombeyo* spp., *Euphorbia* spp., *Hippobromus pauciflorus, Kirkii wilmsii, S. birrea, Ziziphus mucronata*
		• Shrub species: *D. cinerea, Euclea crispa, Grewia* spp., *Ehretia rigida, Aloe* spp.
		• Herbaceous species: *Aristeda* spp., *Eragrostis* spp., *Melanis* spp., *Panicum* spp., *Themeda triandra*
Impacted Vegetation	Impacted Woodland	• Human-utilised areas resulting in poor ground cover, reduced vegetation growth and exposed soil patches
	Impacted Thicket and Bushland	• Associated with rural settlement areas and subsistence farming (i.e. communal lands), where land is intensively utilised for livestock grazing and natural resource harvesting
Settlement	Settlement	• Permanent and near-permanent, formal and informal settlement expanses
		• Garden-plots and farm-holds associated with individual houses
		• Building densities range from low to high and include residential small-holding properties
		• Transport-related infrastructure and structures associated with commercial business
Mines	Mines and quarries	• Surface mining and associated operational infrastructure, including that of underground mining operations
Cultivated land	Cultivated land – Sugarcane	• Areas formally cultivated with permanent (i.e. sugarcane, citrus orchards) and temporary (i.e. maize, wheat) crops.
	Cultivated land – Other	• Large formally delineated subsistence fields
	Cultivated land – Fallow	• Fallow lands and areas being prepared for crops
Forest	Forest – Plantation and Indigenous	• Indigenous forest and commercial forest plantations
		• Commercial plantations are dominated by exotic species (*Pinus, Eucalyptus*) and are distinguishable from natural forest by the systematic planting pattern
		• Indigenous forest refers primarily to the northern Mistbelt Forest Type[14]
Clearfell	Plantation clearfell	• Residual areas after timber harvesting
		• Dominated by grasses, pioneer species, exotic invader species (e.g. *Solanum mauritianus*) and opportunistic seedling recruits of plantation species
Grassland	Grassland	• Natural areas dominated by non-woody plant communities: grasses (Poaceae) and other herbaceous vegetation
Burn Scar	Burn scar	• Recently burnt areas
Cloud Cover	Cloud cover	• Unavoidable cloud cover

Source: Adapted from Thompson[23]

and rural characteristics) population resides (more than 90% of a regional population of more than two million, which, in places, exceeds densities of 300 people/km^2)[16]; poverty and poor economic opportunities have ensured a heavy dependence on the local environment to supplement household income.[17,18,19]

Data sources

Annual status quo land-cover maps were created from Landsat TM images for the period 1993–2006. The pixel resolution of Landsat imagery is fairly coarse (30 m) when compared with other satellite imagery (e.g. SPOT-5: 2.5 m – 10 m, QUICKBIRD: 2.4 m, IKONOS: 1 m – 4 m), as a result of, (1) the large spatial extent of the study region, (2) the temporal extent of the analysis period and the need for annual data and (3) the ultimate purpose of the remote-sensing exercise (i.e. to develop a threat layer for use in a conservation plan). Nevertheless, Landsat imagery was deemed most appropriate for this study because, for prediction purposes, the long-term nature of this dataset is especially valuable. Apart from the analytical benefits of multiple snapshots of the landscape over time, the 13-year analysis period provides an awareness of the change dynamics in this landscape. It incorporates the advent of significant social change in South Africa's recent history (since the 1994 elections) – social change that is likely to have far-reaching consequences for land-cover transformation (e.g. land reform and development proposals in the poverty nodes).[8,12]

For this paper, only the years reflecting the beginning and the end of the analysis period were compared. Winter images were selected to ensure minimal cloud-cover, matching the anniversary of image acquisition as closely as possible to reduce seasonal effects. The raw images were acquired from the Satellite Applications Centre of the Council for Scientific and Industrial Research (http://www.csir.co.za/SAC/). These data were preprocessed by the Remote Sensing Research Unit, at the Meraka Institute (http://www.meraka.org.za), using open source software for high-speed image processing and registration methods.[20]

Land-cover maps were created by a supervised classification method using a maximum likelihood function in which equal prior probabilities were assigned to cover classes. Training sites were based on pre-existing knowledge of the landscape and were selected to best represent each cover type. Classification maps were produced and analysed using geographic information system (GIS) software packages ArcMap 9.2 [21] and Land Change Modeler for ArcGIS[22]. Changes over the intervening 8 years (for which we have data) were mostly linear, although there was some transient oscillation, and generally reflect the net changes discussed in this paper. All analyses excluded the KNP and adjacent private PAs (now within the KNP fence line) because, as a result of the KNP's permanence and statutory protection, it remains committed to conservation, regardless of any land-cover changes occurring within its borders. Any land-cover changes that do occur within the KNP's borders are unlikely to be the result of extensive human activity, but rather a result of more natural drivers. Thus the inclusion of this area into a land-use 'threat layer' is unnecessary.

Cover classes reflect broad land-cover types specific to the development of a threat layer for conservation planning purposes (i.e. generic conservation priority cover types and land-use threats across the BR, Table 1). The classification structure is hierarchical to a degree, but fine-scale vegetation types were not mapped. Where possible, the naming conventions of the National Land-cover (NLC) classification scheme were followed.[23]

Although 'land-cover' and 'land-use' are related, the terms cannot be used interchangeably. Land-cover refers to the features that cover the earth's surface, whilst land-use involves the human utilisation of a specific land-cover or landscape unit.[23,24] While there can only be one land-cover associated with a specific point at a particular time, a particular land-cover can have multiple land-uses. The classification scheme proposed here reflects land-cover, indicating those major cover types that are specific to the development of both a priority layer and threat layer for the BR. Similarly, the map legend differs from the land-cover classification scheme, in that it presents the land-cover classification in a hierarchical manner, indicating the main cover types applicable to the study objectives (conservation plan) and specific to the scale of the study region[24] (BR scale).

Data analyses

The accuracy assessment of the classification maps was a three-stage process. Stage 1 involved ground-truthing, in which we compared classified maps with the national land-cover maps (1994 and 2000) and original satellite images for major discrepancies (e.g. forest within wetland areas). Although it is unlikely that any remotely sensed land-cover map (including the NLC maps) is a perfect description of reality,[25] this step served to improve the initial supervised classification. If unlikely classifications existed, those pixels were re-examined and adjusted if necessary.

Stage 2 ground-truthing made use of historical aerial photographs, as well as site visits and GPS points for more recent images, while Stage 3 involved post-classification expert refinements of classified images that were based on the outcomes of Stages 1 and 2. For each year, the agreement between classified maps and the real-world was assessed using Cohen's Kappa statistic (K) for classification accuracy[26]; K was calculated on a class-level basis, as well as at the landscape level using the combined class-level ground-truth points (GTPs, Table 2). The number of GTPs per class were initially area-proportional, but subsequently depended on the classification difficulty and potential for spectral confusion between specific cover classes (e.g. Settlement versus Impacted Vegetation, owing to the scarce ground cover and exposed areas in both classes). The strength of classification agreement can be interpreted using accepted benchmarks[27] (Table 3), but, for the purpose of this study, $K > 0.75$ was deemed satisfactory.

Analysis years were classified independently, as were the classes within analysis years, to avoid compounding classification errors through the time series. This, together with the high Kappa values, enhanced the confidence that can be placed in the status quo maps and the overall change analysis.

TABLE 2
Class-level and landscape-level classification accuracy of status quo land-cover maps (1993 and 2006)

Class	1993	2006	Points per class
Settlement	0.867	0.854	250
Impacted Vegetation	0.868	0.772	250
Intact Woodland	0.871	0.834	250
Intact Thicket and Bushland	0.808	0.869	250
Grassland	0.916	0.911	100
Forest	0.906	0.942	200
Clearfell	0.846	0.978	100
Cultivated land	0.900	0.800	150
Mines	0.974	0.968	100
Water	0.907	0.964	70
Landscape total	0.802	0.782	1720

TABLE 3
Interpretation of Cohen's Kappa statistic

Kappa statistic	Interpretation: Strength of agreement
< 0	Poor
0.0–0.20	Slight
0.21–0.40	Fair
0.41–0.60	Moderate
0.61–0.80	Substantial
0.81–1.0	Almost perfect

We quantified the net change in land-cover classes between 1993 and 2006 as the percentage change relative to the original 1993 extent. The change is expressed spatially, highlighting areas of transitions between particular cover classes, as well as areas of persistence over the analysis period.

The accuracy assessment of the extent of change compared class-specific classification errors to the magnitude of the changes that were observed. Should the classification error be greater than the observed change, the change reported may be an artefact of pixel misclassification, rather than a genuine change in the landscape. The formula for calculating the classification error was: (1 - the observed agreement of GTPs and status quo maps) × 100.

The formula we proposed above for assessing the extent of change, only considers agreement between GTPs and the corresponding cover class and is expressed as a percentage. In contrast, Cohen's Kappa considers agreement (true positive and true negative predictions), Type I errors of commission (false positives), as well as Type II errors of omission (false negatives), through the use of a confusion matrix and is expressed as a value between $0 \leq K \leq 1$.

RESULTS

Classification method

Given that class-level Kappa statistics indicated a 'substantial' to 'near perfect' agreement with the real world (Table 3), the portrayal of the K2C landscape by the status quo maps was viewed with confidence. Similarly, as the classification error was less than the observed change (Table 4), class-level changes were deemed valid. However, the range of classification errors (0.2% – 5.2%) suggested a differential rate of misclassification across cover types. This range was a function of the class-level classification difficulty and the potential overlap between the spectral signatures that defined each cover-type. Cover-types that displayed within-class heterogeneity, and those that had the potential to share similar characteristics (e.g. Intact Woodland and Intact Thicket and Bushland; Settlement and Impacted Vegetation), suffered a degree of spectral 'confusion' in certain areas. This resulted in a relatively higher incidence of pixel misclassification, as reported by the higher classification errors (e.g. those within the Impacted Vegetation and Intact Vegetation classes, as well as within Settlement areas). Classes with more homogenous cover and distinctive spectral signatures

South African Journal of Science Article #221

displayed less misclassification and, as a result, produced lower classification errors across the class expanse (e.g. Mines and Water).

General land-cover changes

Overall, 36% of the subregion underwent a transition from one cover class to another between 1993 and 2006, regardless of the directional nature of that transition (Figure 2). Spatially, land-cover change dominated in areas of significant anthropogenic activity, that is, in the densely populated rural communities, as well as in the agricultural hubs (e.g. around Hoedspruit). The plantations in the southern limb were also inherently dynamic. The combination of afforestation-harvesting and burn cycles resulted in spatio-temporal inconsistencies in felling across the forestry areas and ensured that these plantations had a fragmented 'persistence' status. In addition, the 2000–2002 construction of the Inyaka Dam immediately west of the plantation area in Bushbuckridge Municipality (Figure 4) further contributed to the change observed at this plantation–settlement interface. The construction of this dam increased the amount of standing water in the BR by 37.1% over 13 years (Table 4).

The protection offered by the private reserves in the central BR contributed considerably to the persistence of the intact vegetation in the subregion, particularly the Woodland-cover class. However, across the escarpment area in the west, topography, and resultant inaccessibility of certain areas, may also have played a role in ensuring the persistence of the Intact Thicket and Bushland cover.

Changes in priority conservation classes: Intact Natural Vegetation

A significant proportion of the BR is dedicated to the 'priority conservation classes' (Figures 3 and 4) – the Intact Natural Vegetation cover classes: Intact Thicket and Bushland (predominating across the escarpment area, where it is interspersed with grassland patches) and Intact Woodland (predominating in the central area).

Between 1993 and 2006, these priority conservation classes declined by a collective 7.3% (an area greater than 460 km^2, Table 4), or 6.5% when natural grasslands were combined into this category, and suffered both 'block' losses (large areas of conversion) and smaller-scale fragmentation to other cover classes. Block losses occurred at the base of the escarpment in the Maruleng Municipality and, most dramatically, on the outskirts of the Bushbuckridge Municipality, extending outwards towards the adjacent privately owned PAs. In this south-central part of the BR, it was the Intact Woodland cover class that underwent the majority of the conversion, changing to a human-impacted version of the original intact vegetation. The Intact Thicket and Bushland class also suffered significant block losses, in the area adjacent to the Bushbuckridge Nature Reserve.

Although there was an overall decline in the expanse of Intact Vegetation within the BR, class-level changes within the combined Intact Vegetation provided interesting results. Across the K2C, Intact Woodland declined by 27.2%, a significant loss of more than 1100 km^2, while Intact Thicket and Bushland increased by 30.2% (approximately 670 km^2). These changes suggested that there were transitional changes within the Intact Vegetation classes (i.e. changes between Intact Woodland and Intact Thicket and Bushland), most likely a result of bush encroachment. Yet, in terms of conservation planning principles, if change is inevitable, changes between intact vegetation classes are preferred over the conversion to other land-cover types, as the resultant cover remains within the 'priority conservation' category.

Changes in Settlement and Impacted Vegetation classes

Over the 13 years, settlement areas increased by 39.7% (more than 180 km^2, Table 4), mainly in the rural areas (Figures 3 and 4), at the base of the escarpment in eastern Maruleng and, especially, across Bushbuckridge. Not only have settlement areas increased in size, but individual settlements have become denser, suggesting a more intensive utilisation of the settlement-space by the (growing) local population. In both municipalities, pre-existing settlement areas expanded outwards, often linking smaller settlement areas and forming areas of near-continuous settlement expanse. This pattern was especially true for areas along transport routes. In Bushbuckridge, a railway line bisects the municipality east to west and village expansion followed this route. However, the smaller, more remote villages in the east also increased, becoming less isolated or developing indistinct village boundaries.

Human-Impacted Vegetation increased from the original 1993 expanse by more than 6.8% (an area greater than 120 km^2) as a result of rural settlement increases (Table 4), suggesting a dependency of the rural communities on the adjacent communal lands.[18,28] Almost all conversion to this cover type has occurred in and around settlement areas, expanding towards the private reserves.

Cultivated land

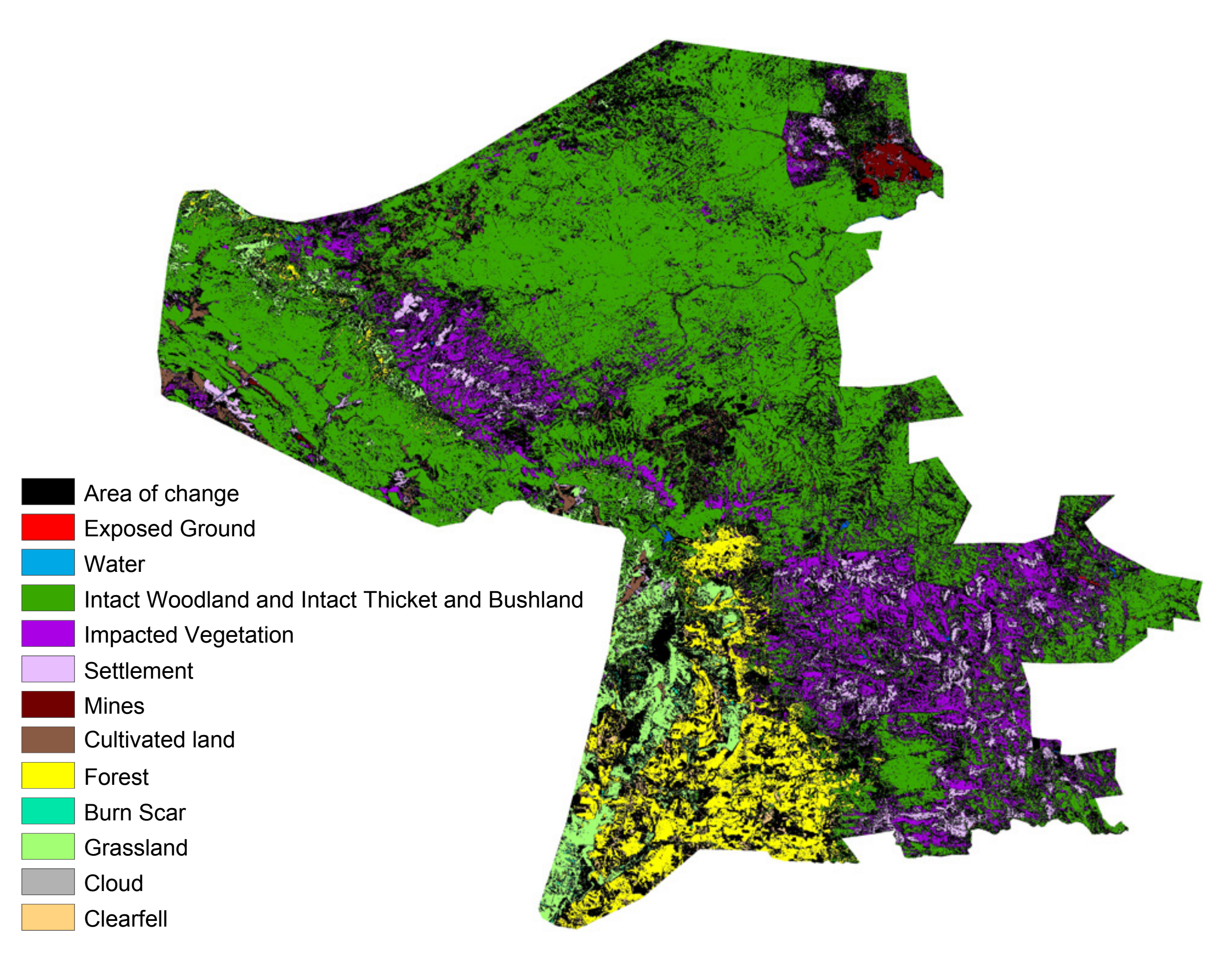
Agriculture in the BR is dominated by commercial farms, the majority of which occur in the central portion, adjacent to Hoedspruit (Figure 4). Other commercial agricultural areas are found along the Ga-Selati River at the interface of the Maruleng – Lepele-Nkumpi – Greater Tzaneen Municipalities and, to a lesser degree, along the Klaserie and Motlatse Rivers. In these areas, the retention of crops during the winter increased by 51.9% in 2006, while fallow areas doubled, resulting in a combined increase of 86% in formal cultivation (Table 4). This value does not reflect changes in subsistence cultivation in the communal rangelands; changes in informal farm-holdings and garden-plots were associated with the human-impacted natural vegetation around settlement areas.

TABLE 4
Percentage change per cover class including class-level classification error for 1993 and 2006

Class	2006 change relative to original 1993 extent (%)	Observed agreement (%) with GTPs		Classification error (%)	
		1993	2006	1993	2006
Urban	+ 39.7	97.2	96.8	2.78	3.13
Impacted Vegetation	+ 6.8	97.0	95.0	2.98	4.99
Intact Woodland	- 27.2	97.1	96.3	2.87	3.63
Intact Thicket and Bushland	+ 30.2	97.5	96.9	2.47	3.07
Combined Intact Vegetation	- 7.3	95.8	94.7	4.12	5.24
Grassland	- 3.7	99.0	99.0	0.92	0.93
Forest	- 2.0 (entire); - 7.1 (southern limb)	98.2	98.9	1.71	1.09
Clearfell	+ 23.7	98.4	99.2	1.58	0.76
Cultivated land	+ 51.9 (with crop); + 86.0 (total)	95.8	96.9	4.14	3.06
Mines	- 6.3	99.7	99.6	0.27	0.33
Water	+ 37.1	99.3	99.7	0.67	0.28

The plus and minus signs respectively indicate an increase or decline in a particular cover.
GTPs, ground-truth points.

South African Journal of Science
Article #221



Areas in black have undergone a transformation to another class. These changes exclude any transitions between the Intact Woodland and the Intact Thicket and Bushland classes. For the purpose of this map, these two classes were combined into a single Intact Vegetation class – regardless of the transitions between the two, they remain a priority class for conservation planning.

FIGURE 2
Net persistence of land-cover classes in the Kruger to Canyons Biosphere Reserve from 1993 to 2006

Mining

The K2C includes the largest opencast mine in southern Africa – the Palabora mine in the north of the BR (Figures 3 and 4). There are also smaller mines elsewhere in the BR, but none of which approach the size of Palabora. The Mines class decreased by 6.3% of the original 1993 expanse (more than 5 km^2) across the subregion (Table 4), suggesting that the surface mine expanse has remained fairly consistent for the study period; consistency of this form is typical of established mining operations.

Plantations and indigenous forests

There was a 2% decline in the original 1993 extent of forests across the subregion. The southern limb includes an extensive portion of Mpumalanga's commercial forestry plantations (mostly pines and eucalypts), as well as patches of indigenous forest. In this southern limb, the Forest class declined by 7.1% (Table 4), while Clearfells increased by 23.7%.

DISCUSSION

The K2C is a region of varied and sometimes seemingly antagonistic land-uses, with the KNP, primary production sectors and poor rural communities juxtaposed across the landscape. Yet it is the combination of these land-uses and associated values that has ensured that the subregion has achieved its BR status. The essence of the BR concept is an enduring human–environment relationship,[15] a departure from the traditional 'fortress conservation'[29] views, which were, until recently and are often still, held by many PAs. Hence, land-cover change in this BR is actually expected. Rather than preventing human usage of the landscape, BRs recognise that conservation initiatives may be a means of enhancing local livelihoods[30] and, thereby, a means of facilitating the sustainable use of resources. The K2C mission statement reflects the overarching Man and Biosphere theme, highlighting biodiversity conservation, whilst ensuring that the development needs of other stakeholders in the subregion are met (see http://www.Kruger2Canyons.com). As such, preferential conservation initiatives in the BR are those that incorporate some form of socio-economic benefit for the wider BR community; especially given the current 'Presidential poverty node' status of two of the region's local municipalities, Bushbuckridge and Maruleng.[12]

The K2C is a coupled social-ecological system, that is, an ecological system intrinsically linked to many different socio-economic elements of the landscape.[31,32] The majority of the people in these communities rely heavily on the natural resource economy[19] that is provided by their immediate environment, both as a supplement to household income and a 'safety net' in times of increased hardship.[18,33]

Change in settlement and the rural cultural landscape

Land-cover change predominates in the areas adjoining the dense rural settlements and a broad footprint of change propagates outwards from the village districts (Figure 2).

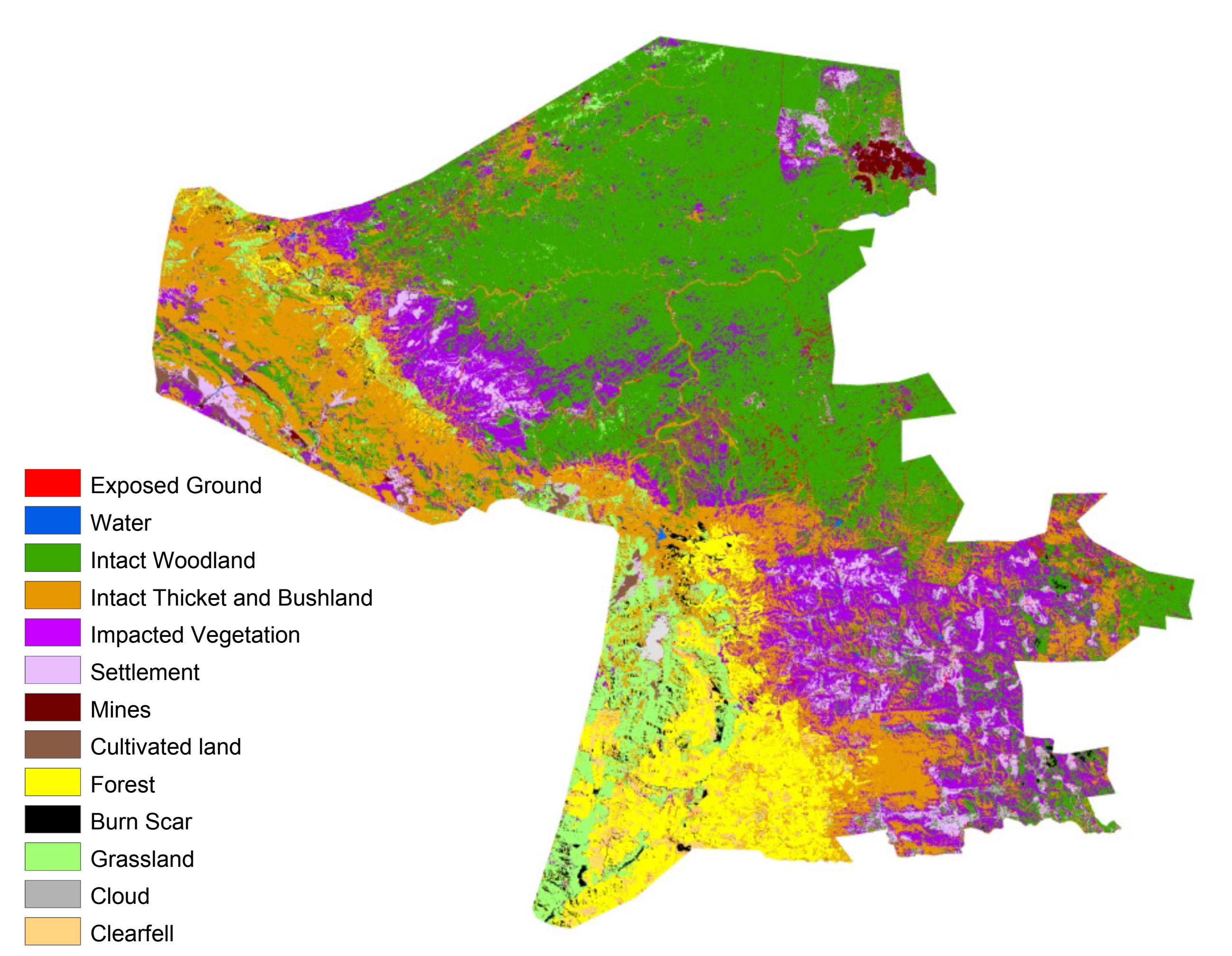


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FIGURE 3

Status quo land-cover map for the Kruger to Canyons Biosphere Reserve (1993)

The former apartheid homelands, with their history of forced settlement, and the more recent influx of foreign refugees into the subregion,[16,34] have produced high human population densities and entrenched poverty.[28,33] The activities of these high densities of impoverished people constitute a primary driver of landscape change observed during the analysis period. In these rural areas, individual villages have expanded and become denser and smaller settlement areas are being near-amalgamated into neighbouring settlements. In particular, settlement expansion tracks transport routes (Figures 3 and 4). In addition, transfers of water to Bushbuckridge from Inyaka Dam may fuel future socio-economic development in the area[35,36] and, as a result, this may have as yet unseen effects on the growth of the regional settlement expanse and, ultimately, land-cover change in the area.

With the expansion of settlement areas, there is a concomitant effect on the surrounding natural environment and the footprint of human impact associated with each village radiates outwards, depleting environmental resources. An analysis of land-cover change (1974–1997)[32] in three villages within Bushbuckridge, also revealed extensive growth of human settlements and an associated decrease in woodland cover. However, the long-term spatial footprint and usage intensity of the communal areas is dynamic[32] and linked to the regional economic climate, local population size, site-specific poverty circumstances and the ecological condition of the environment being utilised. As such, these communal areas pulse with the regional socio-economic conditions and may alter in times of stress.

While expansion of settlement and rangeland are intrinsically linked, the rate of settlement expansion exceeded the rate of rangeland growth during the analysis period. This observation suggests that the spatial limits imposed on the communal areas by surrounding land-uses (i.e. existing village and PA distribution) may prevent the equivalent spatial expansion of rangeland in response to settlement increases. Should this trajectory continue, settlements will continue to erode communal areas, potentially resulting in an unsustainable use of the natural resource base.

In terms of conservation planning priorities, a significant proportion of the Intact Woodland and Intact Bushland loss occurred around these dense settlement expanses (Figures 3 and 4). Here, large areas of this intact vegetation were transformed to an intensively utilised cover type (e.g. around the Bushbuckridge Nature Reserve). Intact Vegetation suffered losses throughout the BR, but from a conservation planning perspective, it is these block losses (as opposed to smaller-scale fragmentation) that are most concerning. Block losses affect coarse-scale structural and functional connectivity and composition,[37,38] potentially undermining conservation possibilities and long-term landscape functioning.

However, change in the study region is not only driven by dense populations of the rural poor. The literature cautions against the assumption of a single, linear change driver (i.e. the expanding rural landscape).[32,39] Drivers of change are likely to be distributed differentially (spatially, temporally, compositionally) across the BR. The industrial sector, the diverse management strategies of

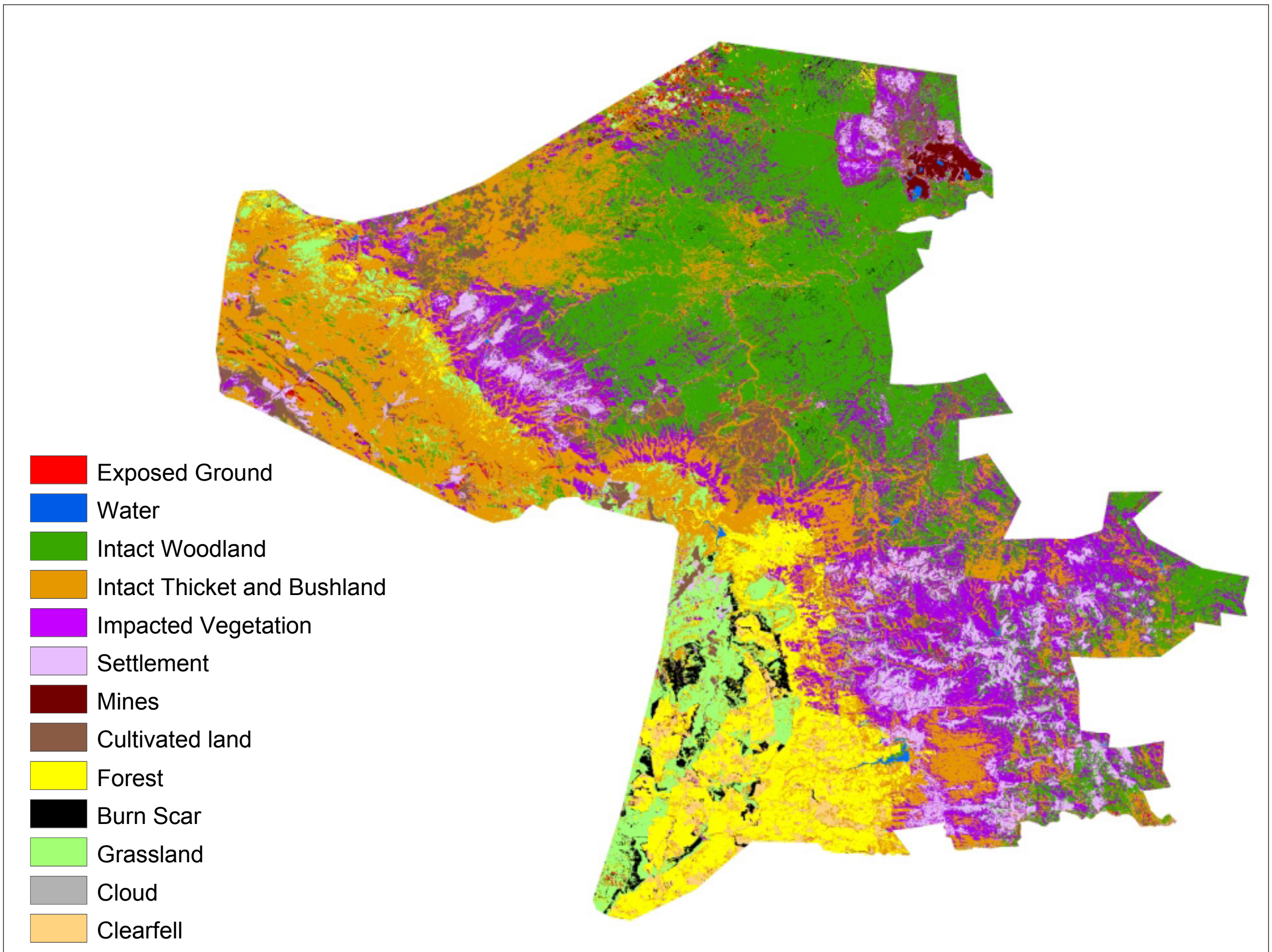

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FIGURE 4
Status quo land-cover map for the Kruger to Canyons Biosphere Reserve (2006)

individual private PAs and the inherent heterogeneity of African savannahs are also responsible for variation in land-cover observed in the BR. For example, the intensive management of the local landscape[40] within private PAs, may resemble areas within communal land, due to the addition of artificial water points,[41] mowing and bush-clearing (Figures 3 and 4). The presence of these bare areas in PAs is detected as land-cover change, as sharply defined as that within intensively utilised communal areas, raises the question of whether all PAs should automatically be considered as 'Intact' and, unequivocally, included in biodiversity planning studies. However, in terms of this study, and in relation to general conservation planning principles[9] (regardless of any ongoing change in the PAs), these PAs will be regarded as a regional conservation commitment, provided that their 'protected' status is maintained.

Change in the core industrial sectors

Commercial agriculture and plantation forestry practices drive considerable change in this landscape (Figure 2), particularly in the Hoedspruit area (agriculture) and in the southern limb of the K2C (forestry). These two sectors are inherently dynamic (agriculture more than forestry) due to the replanting–harvesting cycle.

Plantation forestry is a significant driver of the formal economy in the BR, particularly in the southern limb, where the extensive Komatiland plantations extend beyond the K2C borders. Given the vast expanse of these commercial plantations, any future changes in related land-cover should have implications for biodiversity conservation in the subregion. However, potential changes in this plantation expanse may be positive – as in the proposed transfer of 15 600 ha of the consolidated Limpopo–Mpumalanga forests to the Blyde River Canyon National Park.[42] After rehabilitation, a portion of these in the Sand River catchment will be used to link the smaller Blyde Reserve with the KNP,[43] effectively enhancing overall connectivity in the BR.

Formal agriculture has increased considerably in the BR during the analysis period. This finding is corroborated by provincial agricultural census trends for the 1993–2002 period.[44] In the 13-year study period, the area contained within the large, formal fields (under crop or fallow, commercial or subsistence) has increased substantially. Crop agriculture in the subregion is distinctively seasonal, but was less so in 2006, when more crops endured through the winter. This pattern may reflect either a change in the method of irrigation since 1993, or increased investment in field crops in the region.[44] Future agricultural developments, such as the biofuel development planned for Hoedspruit, may be important for the BR, with significant implications for future land-cover change.[45]

Although mining occupies a relatively small portion of the total available land-surface in the BR (less than 5%), environmental consequences of mining are often locally severe. At this time it seems likely that operations at Palabora mine may continue for at least 20 years, but its eventual closure may have significant repercussions for the regional social-ecological system (i.e. the generation of 'new-poverty'[46] within the mine-employed community after mine closure). Reliance on the rural resource economy may therefore increase, with negative effects on settlement–communal land relationships and regional land-cover change.

RECOMMENDATIONS AND CONCLUSION

Land-cover change reflects both the status of economic development and the landscape character and condition[47] and will vary with future socio-ecological circumstances. The identification of historical drivers and the possibility that existing trends will continue into the future may have important implications for biodiversity protection in the K2C. In this paper, we have quantified land-cover changes in this landscape, while speculating as to the drivers of this change, yet the long-term transition probabilities between cover classes still need to be determined in a realistic context. Directing conservation action at areas that will remain intact and secure from future transformation, ensures the best use of limited conservation resources, and promotes enduring biodiversity protection.

In terms of application within a conservation planning framework, land-cover data provide an essential 'threat' layer and a spatial basis to determine the potential land-cover conversion risk of important biodiversity features. Yet land-cover data are but one of the required layers of a conservation plan. Economic cost surface data and biodiversity data are also required for a conservation plan, which needs to integrate a wide range of data sets to be effective. Land-cover data implemented within this framework provide a systematic process that addresses the spatial requirements of both conservation and other stakeholders, attempting to reconcile conservation land-use with other land-uses in the BR.

Given the lag between the accumulation of historical land-cover data and the development and, ultimately, the implementation of the conservation plan,[48] it is critical that the methodology employed allows for additional land-cover data sets to be added to the time-series. The raw satellite data are readily available, the procedure used to translate raw images to usable land-cover data sets is clearly defined and, as a result, repeat procedures are feasible and easily achieved. Although this land-cover data set was created for use within a conservation planning framework, these data have multiple applications for PA managers (e.g. in regard to management decisions), community conservation and sustainable resource-use initiatives (e.g. in relation to issues of sustainable harvesting), land-use planners (e.g. regarding settlement growth and associated service delivery) and researchers alike. We hope that these data, and the subsequent studies that develop from their use, will make a valuable contribution to the stakeholders present in the subregion.

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Chapter 4

The Race for Space: Tracking land-cover transformation in a socio-ecological landscape

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The Race for Space: Tracking Land-Cover Transformation in a Socio-ecological Landscape, South Africa

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Abstract Biosphere Reserves attempt to align existing biodiversity conservation with sustainable resource use, specifically for improving socio-economic circumstances of resident communities. Typically, the Biosphere Reserve model is applied to an established landscape mosaic of existing land uses; these are often socio-ecological systems where strict environmental protection and community livelihoods are in conflict, and environmental degradation frequently accompanies "use". This raises challenges for successful implementation of the model, as the reality of the existing land-use mosaic undermines the theoretical aspirations of the Biosphere concept. This study focuses on the Kruger to Canyons Biosphere Reserve (K2C), South Africa; a socio-ecological landscape where formal conservation is juxtaposed against extensive impoverished rural communities. We focus on land-cover changes of the existing land-use mosaic (1993–2006), specifically selected land-cover classes identified as important for biodiversity conservation and local-level resource utilization. We discuss the implications of transformation for conservation, sustainable resource-use, and K2C's functioning as a "Biosphere Reserve". Spatially, changes radiated outward from the settlement expanse, with little regard for the theoretical land-use zonation of the Biosphere Reserve. Settlement growth tracked transport routes, transforming cohesive areas of communal-use rangelands. Given the interdependencies between the settlement population and local environmental resources, the Impacted Vegetation class expanded accordingly, fragmenting the Intact Vegetation class, and merging rangelands. This has serious implications for sustainability of communal harvesting areas, and further transformation of intact habitat. The distribution and magnitude of Intact Vegetation losses raise concerns around connectivity and edge effects, with long-term consequences for ecological integrity of remnant habitat, and K2C's existing network of protected areas.

Keywords Biodiversity conservation · Fragmentation · Kruger to Canyons Biosphere Reserve · Land-cover change · Landscape connectivity · Man and the Biosphere Programme

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Introduction

Given the spatial discrepancies between habitat protection, habitat transformation, and subsequent degradation (Rodrigues and others 2004; Scholes and Biggs 2005; Hoekstra and others 2005), resurgent protectionists began to advocate more strongly for the return of fortress conservation (reviewed in Wilshusen and others 2002; Holt 2005). However, with concessions to social justice and equality issues around resource management, contemporary conservation has had to become more socially, politically, and economically accountable.

As far back as the 1980s (viz. 1982 World Congress on National Parks), there was consensus that conservation efforts would only endure if they considered human needs (Naughton-Treves and others 2005). Yet it is only more recently that sustainable utilization of the protected environment has been revisited in biodiversity management

(Brown 2002; Zimmerer and others 2004), with the notion that sustainable economic development need not result in the degradation of broader ecological values.

The Biosphere Reserve (BR) Model of UNESCO's Man and the Biosphere (MAB) Program is one of the first international conservation designations, which stresses the complementarities and tradeoffs between conservation and development, rather than conflicts between traditionally competing land-use types (Brown 2002), i.e., the protection of biodiversity while encouraging its sustainable use (UNESCO 1996; Nolte 2008). BRs are hailed as opportunities to reconcile traditional exclusionary biodiversity conservation with development (UNESCO 1996), for the specific benefit of enhancing the socio-economic circumstances of communities living within the BR-landscape on a sustained basis.

Since inception in 1976, the BR model has been especially well received by both developed and developing nation governments, i.e., the notion of making conservation "work for the poor". The rate of buy-in into the Program has increased steadily, escalating rapidly after 1995 when the Seville Strategy refocused the core objectives on sustainable development (reviewed in Coetzer and others 2013). Internationally, there have been a number of successful economic development initiatives implemented as a result of the MAB Program, with development projects enhancing both economic circumstances and local ecological integrity (Coetzer and others 2013 includes a comprehensive list). Although the BR concept of "dual benefit for development and conservation" is theoretically very attractive, the practical implementation is challenging in many circumstances, as the model is usually applied to an already well-established landscape mosaic of existing land uses. This land-use mosaic, and its spatial land-cover change dynamics, underlies the successful implementation of the BR concept in any particular landscape.

In the socio-ecological systems that typify BR placement, biodiversity conservation and community livelihoods are often in conflict. Maximizing one goal is frequently to the detriment of the other (Cortina-Villar and others 2008), and in many developing countries in particular, the need to access environmental resources to alleviate poverty is more pressing than longer-term environmental conservation (Mwavu and Witkowski 2008). The reality of the socio-ecological system is frequently one of competing aspirations for well-being; a balancing act between ecological integrity and associated ecological values, broader socio-economic development and political goals, and the self-determination of local stakeholder communities (e.g., over resource use) (Smith 2003).

Thus, a complicated combination of ecological resources and social/economic/political (development) factors, both past and present, governs the spatial distribution of the land-use mosaic, which in turn determines the effectiveness of individual BRs. Understanding these links is critical to manage biodiversity sustainably (Sivrikaya and others 2007) and minimizing further losses of biodiversity, with both the location of land-cover change and the "quantity" of land-cover change (i.e., the demand for resources determining land-use patterns and fueling conversion) shaping the trends in landscape transformation (Trisurat and others 2010). The ecological consequences of landscape transformation are tied to the loss and fragmentation of natural cover (with consequences for habitat connectivity) and the homogenization of landscape variation (Fahrig 2003; Fischer and Lindenmayer 2007; Verburg and others 2012). The degree to which these result in degradation and loss of ecological function from the landscape will depend on the dynamics and spatial configuration of remnant patches (Cakir and others 2008), and the ability of managers to identify unfavorable change trajectories and pre-empt large-scale patterns of further loss.

The Kruger to Canyons Biosphere Reserve (K2C) in South Africa is the focus of this study. It includes the iconic South African protected area, the Kruger National Park (KNP) and a vast array of smaller, privately-owned conservation areas that are internationally recognized as superior tourism destinations. These are juxtaposed against an impoverished rural population engaged in subsistence livelihoods, as well as regionally important economic sectors, e.g., copper mining, plantation forestry, commercial citrus, and sugarcane agriculture.

The BR designation here has been applied to a long-established socio-ecological system, with the hope of redressing the inequalities in socio-economic circumstances in the subregion. Yet the reality of the existing land-cover/land-use mosaic may have little regard for the nomination of "Biosphere Reserve", particularly if, (1) the restrictions on resource use implicit in the BR designation compromises existing livelihoods and resource utilization (i.e., discussed in Coetzer and others 2013), or (2) the development aspirations of the model degrade the ecological values of this world-renowned conservation landscape. Thus, understanding the mosaic is critical to the effectiveness of the BR model as it has currently been applied to this subregion.

The aim of this study is to examine the underlying land-use mosaic on which the success of K2C depends, with the particular goal of assessing land-cover changes linked to the relationships between important societal, economic, and biodiversity factors in this socio-ecological landscape. The data we present here, and the approach that we set out, allow us to understand the drivers of land-cover change in K2C, interpret the spatial dynamics of this change, its effects on existing and future biodiversity conservation, as well as for the longer-term sustainability of the BR model in this, and other, extensively utilized landscapes.

We analyze the spatial nature of net land-cover change in K2C (1993–2006) with regard to the theoretical zonation (core, buffer, and transition) of the BR model, and the existing ecological values in this landscape. We focus on consolidated land-cover classes identified in a previous study (Coetzer and others 2010) as important for biodiversity conservation and local-level resource utilization here. We assess the land-cover transformation patterns of these consolidated classes, specifically fragmentation patterns, using landscape metrics to quantify patterns of change. We relate the implications of these patterns to further fragmentation and degradation of K2C's intact habitat, as well as to future conservation and management options for the BR, specifically the validity of MAB BR model for this subregion.

Methods

Study Site

Approximately 2.6 million ha in extent, the Kruger to Canyons Biosphere Reserve (K2C, http://www.kruger2canyons.org; Fig. 1) is one of largest BRs in the world, and the largest in South Africa. Located in the north-eastern part of South Africa (Fig. 1b), K2C includes three biomes: Savanna, Grassland, and Forest (Mucina and Rutherford 2006), further divisible into associated vegetation units (*Bioregions*): Combretum Bushveld, Lowveld, Mesic Grassland, and Mopane Bioregions, with their original, pretransformation extent (as opposed to extant extent) indicated (Fig. 1c).

Established in 2001 by UNESCO's Man and the Biosphere (MAB) Program, K2C includes an extensive system of both national and privately-owned protected areas (PAs), dominated by the central portion of Kruger National Park (KNP; full extent ~2 million ha) in the east (Fig. 1c). This vast expanse of conservation areas forms the *core-* and *buffer*-zones of K2C (Fig. 1d); here, biodiversity conservation [National PAs with strict statutory protection in core zones; other conservation designations without statutory protection in buffer zones (e.g., privately-owned or community-managed conservation areas)], and only those land uses compatible with conservation (e.g., ecotourism and recreation) are permitted. Given this spatial investment to a single land-use type, i.e., conservation, land use in the remaining unprotected land, although theoretically and spatially restricted to the *transition* zone, remains intensive.

The zonation of all BRs is intended to ensure a graduated scale of land-use intensity relative to the core zone, preventing the development activities in the transition zone from impacting upon existing conservation areas (Coetzer and others 2013). However, the zonation of this BR is not the stylized concentric rings of model BRs, nor is it the spatial arrangement of core: buffer: transition zone envisioned originally by MAB. Rather, the zonation has had to be applied to an established land-use mosaic, which has implications for spatial buffering between the core and transition zones (Fig. 1d).

In the transition zone, economically important agriculture-, plantation forestry-, and mining-sectors predominate here (Fig. 1c), and the already substantial settlement areas continue to expand (Coetzer and others 2010). The majority of the regional population resides in these settlement areas, mainly the rural and rurban (having both urban and rural characteristics) areas located in former apartheid homelands, and present day "Presidential Poverty Node" municipalities of Bushbuckridge and Maruleng (Fig. 1a). There is a stark dichotomy between proclaimed PAs (core or buffer PAs), with their "big game" attractor and international tourism appeal, and the open access, non-wildlife areas located beyond PA fences. In places, settlement population densities may exceed 300 people/km^2 (Pollard and others 2003), despite being a semi-arid environment. The growing population, poverty, and poor economic opportunities have ensured a fundamental reliance on the natural resource base to augment household income—long-term repercussions of the forced resettlement of black people to these rural "homeland" areas, and the separate development policy of the apartheid regime. Here, the communal rangeland areas associated with settlement units offer invaluable livelihood options for the population of these rural areas, supplementing households through subsistence agriculture, livestock grazing and the direct harvesting of fuel- and construction-wood, food sources, and weaving materials (Shackleton and Shackleton 2002; Dovie and others 2005; Twine 2005).

Data Sources

The 1993 and 2006 land-cover maps were created from winter Landsat^{TM} archival satellite imagery (pixel size: 30 m), matching the anniversary of image acquisition as closely as possible to reduce seasonal affects. In South Africa, winter falls within the dry season (May–August) ensuring lower standing phytomass and plant cover. Thus, winter images ensured minimal cloud cover, and maximum visibility across all scenes, and allowed clear distinction between managed and natural vegetations.

Original land-cover maps showed broad land-cover types pertinent to the development of a land-use layer for K2C (Coetzer and others 2010). For the purpose of this study, the K2C study region excluded KNP and the adjacent private PAs (now within the KNP game fenceline), rather focusing on the matrix area beyond the Park (see

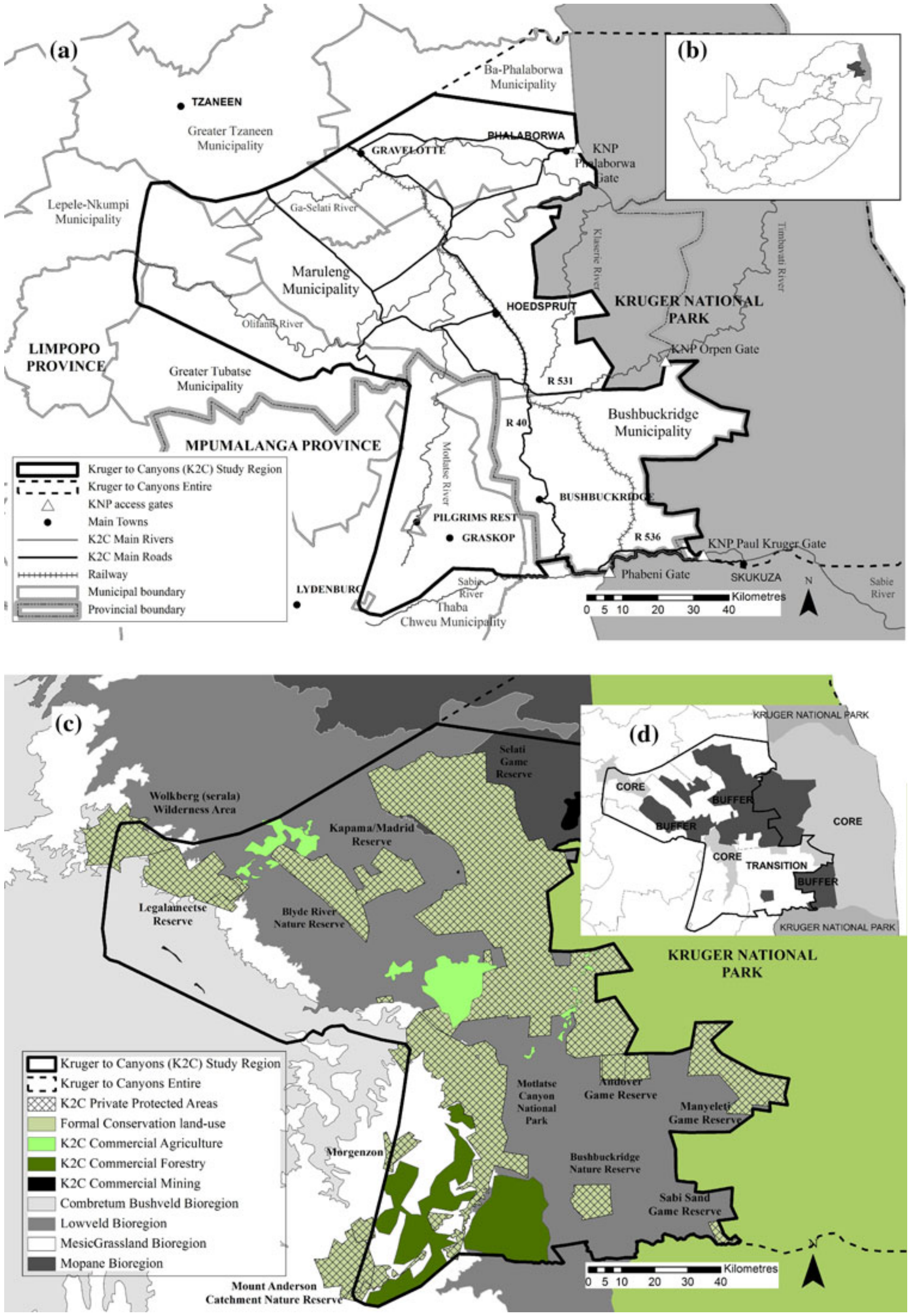

Fig. 1 **a** Regional location of the Kruger to Canyons Biosphere Reserve (K2C), including provincial boundaries, municipal districts, main towns, roads and railway lines, and major rivers. *Inset* (*b*) Map of South Africa showing K2C location relative to national and provincial boundaries. **c** Regional location of the K2C, including land use, privately-owned protected areas, and vegetation bioregions. *Inset* (*d*) The spatial arrangement of core, buffer, and transition zones in K2C

Coetzer and others 2010). Hence, the "core" zone referred to in this article reflects only the *internal* core PAs (hereafter "core zone"), rather than the full extent of internal PAs and central KNP.

The three selected land-cover classes presented here: Intact Vegetation, Impacted Vegetation, and Settlement (Table 1) were extracted, and consolidated, from sixteen classes originally classified in the 1993 and 2006 classification maps. They account for more than 75 % of the classified landscape in both years.

Of these selected cover classes, Intact Vegetation represented a biodiversity conservation class of untransformed habitat, thus representing the "environmental/conservation" aspect of the theoretical BR model (Fig. 2). In contrast, Impacted Vegetation and Settlement classes represented land-cover in varying states of human utilization

Table 1 Hierarchical description of three land-cover classes classified for the Kruger to Canyons Biosphere Reserve (K2C)

Consolidated land-cover classes	Class description (including classification accuracy (*K*) of each class for 1993 and 2006 classification maps; see Coetzer and others 2010 for expanded version)
Intact Natural Vegetation (Fig. 2)	Moderately open savanna; tall–medium sized trees with a distinct herbaceous layer (Intact Woodland: $K_{1993} = 0.871$; $K_{2006} = 0.834$)
	Moderate–dense woodland with a well-developed shrub layer, distinct herbaceous layer [Intact Thicket and Bushland: ($K_{1993} = 0.808$; $K_{2006} = 0.869$)]
	Untransformed by anthropogenic activities
	Includes bush encroached areas—owing to spectral similarities to intact thicket vegetation (Fig. 2c)
	In the original Coetzer and others 2010 classification hierarchy, the forest class was treated separately, here for forest patches located with the PAs (and verified as such), we have combined into the Intact Vegetation class. Specifically, Intact Vegetation areas within the Motlatse Canyon National Park (formally *Blyde River Canyon National Park*) also include small patches of indigenous forest areas known to be found within this PA. Forest *K*(indigenous & commercial): $K_{1993} = 0.906$; $K_{2006} = 0.942$; with commercial forestry in neighboring area, exterior to this PA distinguishable from natural forest by the systematic planting pattern
Impacted Vegetation (Fig. 3)	Human-utilized areas resulting in poor ground cover, reduced vegetation growth and exposed soil patches (Impacted Woodland, Impacted Thicket, and Bushland: $K_{1993} = 0.868$; $K_{2006} = 0.\ 0.772$)
	Associated with rural communal rangelands where livestock grazing, natural resource harvesting, and subsistence farming occurs
Settlement (Fig. 4)	Formal- and informal-settlement area with associated infrastructure ($K_{1993} = 0.867$; $K_{2006} = 0.\ 0.854$)

and modification (Figs. 3, 4), from moderately-extensively utilized (Impacted Vegetation, representing the communal rangeland areas) to completely transformed (Settlement). They fulfilled a "development/human utilization" aspect of the BR model, and represented priority land-cover threat classes, i.e., proxies for the processes that may compromise biodiversity in this landscape in the long term. Together, these two classes represented significant drivers of the change observed in K2C.

The accuracy assessment of classification outputs was a 3-stage, iterative process of classification improvement. Stage (1) involved assessment of broad-scale classification discrepancies with the original satellite images, as well as national land-cover maps (1994 and 2000, on which the naming conventions of cover classes were based), Stage (2) ground truthing of classification outputs comprising ad hoc historical aerial photographs, site visits, and GPS points, and Stage (3) post-classification expert refinements of classified images, including manual digitization of selected areas/cover—based on the outcomes of stage 1 and 2 (comprehensively described in Coetzer and others 2010).

The classification accuracy of the resultant land-cover classes was measured using Cohen's Kappa statistic (*K*, Cohen 1960; Foody 2002), where *K* is expressed within the range $0 < K < 1.0$, rather than a percentage (as in the case of percentage agreement/classification error, see Coetzer and others 2010). *K* values approaching 1 indicate complete agreement between classification maps and reference dataset, while those closer to 0 indicate chance agreement only (Ismail and Jusoff 2008). The well-accepted benchmarks of

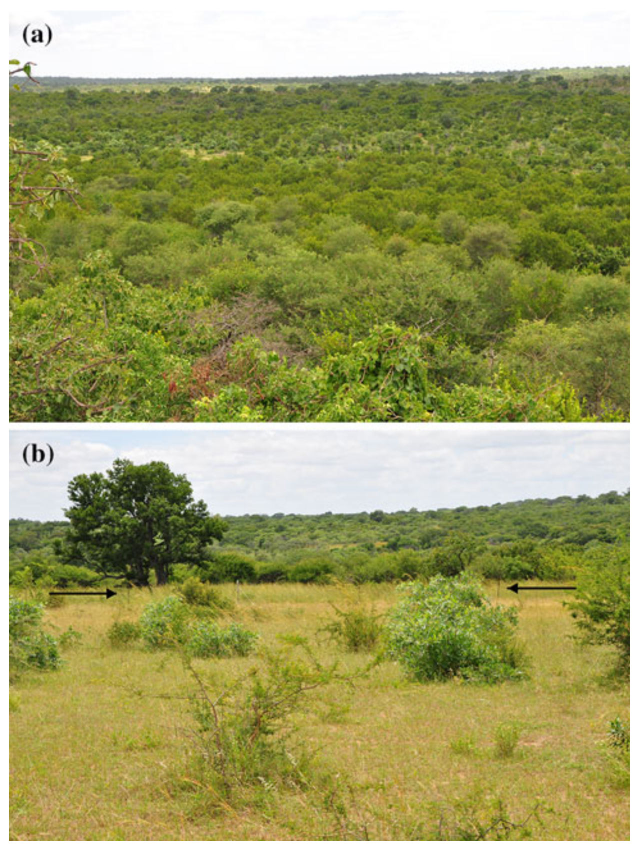

Fig. 2 Intact Vegetation cover class. *Inset* **a** elevated view of Intact Vegetation cover; **b** East-facing fence-line contrast between Intact Vegetation class (Andover Game reserve, background) and adjacent human-utilized rangeland in foreground. *Arrows* highlight fence position. (Photo credit: B.F.N. Erasmus; Andover)

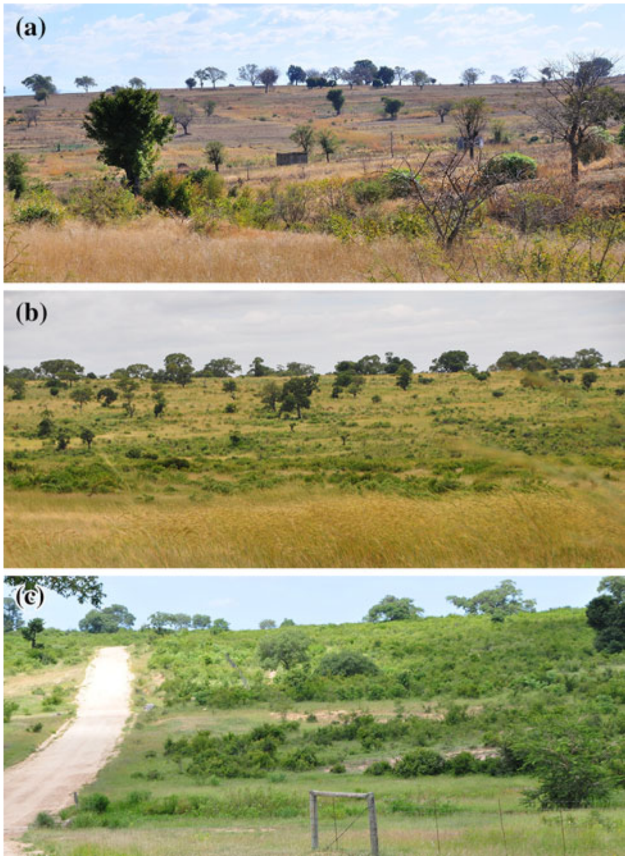

Fig. 3 Impacted Vegetation cover class, showing communal rangeland areas undergoing degrees of environmental utilization: **a** subsistence cultivation with clearly delineated fields, **b** moderately bush encroached areas owing to resource harvesting, which may progress into **c** dense bush encroached area depending on intensity of harvesting. (Photo credit: B.F.N. Erasmus; Bushbuckridge)

Fig. 4 Representative photograph of Settlement cover class in the Kruger to Canyons Biosphere Reserve (K2C); houses are associated with garden plots, cattle in the foreground are driven to the neighboring communal rangeland area for grazing. (Photo credit: B.F.N. Erasmus; Xanthia, Bushbuckridge)

Landis and Koch (1977) provide interpretation of K, as the *strength of agreement* between classification maps and reality (Table 2). For the purpose of this study, $K > 0.75$ for the selected classes (Table 1), demonstrating "Substantial," or "Near Perfect" agreement with the real-world environment.

Data Analysis: Assessing Spatial Pattern

The transformation of each priority land-cover class was assessed using IDRISI's Land Change Modeler (Eastman, 2012) and ESRI ArcMap 10 (ESRI 2010) at the landscape- and BR zone-scale. The patterns of net gains (G; conversion *to* selected cover from another cover type), net losses (L; conversion from selected cover to another cover type), and net persistence (P; areas that no change in spatial extent of selected cover over the analysis period) for each selected cover class were examined through the creation of Gains:Losses:Persistence (GLP) maps from the original 1993 and 2006 land-cover maps. GLP maps indicate *total spatial* change, rather than only net change/spatial distribution as reflected by classification maps. In this context, patches of gains, losses, and persistence occurred within each Intact-, Impacted-Vegetation, and Settlement cover class, where they formed discrete classes of "change," and were analyzed as such, providing insight into the pattern/process of change.

However, changes in distribution, as presented by the 2006 classification map reflect "gains made" and "persistence of" cover classes over time, with: losses of a particular cover class amalgamated into the spatial gains of another class. Similarly, change in the selected classes does not only occur between Settlement, Impacted cover, and Intact cover classes, although we only focus on these class here. Thus, additional classes are briefly mentioned in the results, as a component of the total change Exposed ground, Cultivated land, Natural and Managed grassland; land cover present in the original classification maps, but not addressed in this article (Coetzer and others 2010 for detail).

Categorical classification maps display the landscape as a mosaic of patches, reflecting areas of homogenous/similar environmental conditions at a particular scale (McGarigal and others 2002; McGarigal 2012). Landscape metrics present a summary of this landscape structure (McGarigal 2012), characterizing the spatial pattern (composition and configuration) of specified land-cover classes across a predefined region. Thus landscape metrics used here (calculated using Fragstats Version 4.1; McGarigal and others 2002), further quantified the spatial patterns of the original land-cover maps, and the Gains:Losses:Persistance (GLP) patterns of change at both landscape- and zone-scales for these selected land-cover classes.

While the use of landscape metrics as an analysis tool is abundant in the literature, we acknowledge the ongoing

Table 2 Interpretation of Cohen's Kappa statistic (after Landis and Koch 1977)

Kappa Statistic	Interpretation: strength of agreement
<0	Poor
0.0–0.20	Slight
0.21–0.40	Fair
0.41–0.60	Moderate
0.61–0.80	Substantial
0.81–1.0	Almost perfect

debate about the most appropriate metric for pattern analysis, the reliability of these metrics across scales, and issues of redundancy and correlation between metrics (Riitters and others 1995; Cain and others 1997; He and others 2000; McAlpine and Eyre 2002; Li and others 2005; Cushman and others 2008). The metrics selected were those deemed most suitable for use within the proposed conservation planning framework, particularly with regard to (i) fragmentation of the selected land-cover classes, (ii) changes in land cover relative to BR zonation, and (iii) the future scheduling of management action in response to land-cover change (Table 3). Metric indices should be interpreted as relative comparisons, but the absolute range for interpretation is presented in Table 3.

Results

Settlement

The Settlement cover class (Fig. 5a) accounted for 5.86 % of the K2C landscape in 2006 (Table 4, PLAND), with the majority (93 %) in the transition zone (Table 5, PLAND; Fig. 6e, f). Across K2C, Settlement gains exceeded losses

Table 3 Landscape metrics selected for quantification of the land-cover change occurring in the Kruger to Canyons Biosphere Reserve (K2C) between 1993 and 2006

Landscape Metric	Metric interpretation (after McGarigal and others 2002) and relevance to study objectives and change detection
Percentage contribution to total landscape area (PLAND; Unit: Percentage, $0 < \text{PLAND} < 100$)	Allows for a relative comparison of the magnitude of change in priority land-cover classes. PLAND → 100 when cover class becomes increasingly abundant in the landscape. Changes in PLAND are not changes in current extent relative to original extent (as in original study), but show changes relative to total landscape area at each time period
Area Weighted Mean Patch Size (AWMPS; Unit: Hectare, $\text{AREA} > 0$)	Estimate of average patch size. Patch sizes decline with increased fragmentation, with consequences for species sensitive to area effects, edge effects, with implications for habitat quality. Weighting patches by size better reflect cover-specific sensitivity to disturbance/change processes (Li and Archer 1997), accounting for skewed distribution of patch sizes (Townsend and others 2009).
Number of patches (NP; Unit: Count, $\text{NP} \geq 1$)	Summary of class contribution to the landscape. Increasing fragmentation results in increase in patch number (up until some threshold where no cover remains).
Area Weighted Mean Shape Index (AWMSI; Unitless: $\text{SHAPE} \geq 1$)	Reflects patch shape irregularity relative to a compact ideal. Patches with greater shape irregularity have increased total edge, with potential for more edge effects with adjoining cover types, affecting habitat quality. 'Area weighting' weights large patches higher than smaller ones, correcting for the Perimeter: area ratio's tendency to vary with patch size. SHAPE = 1 when all patches in the landscape are square, increasing as patch shape become more irregular.
Aggregation index (AI; Unit: Percentage, $0 \leq \text{AI} \leq 100$)	AI gives insight into the process of fragmentation specific to a cover type, measuring changes in contagion. Refers to the tendency of a cover class to be spatially aggregated at the smallest (pixel-level) scale. AI = 0 when the class is maximally disaggregated (with no like adjacencies). AI increases with aggregation; AI = 100 indicates a single, compact patch (not necessarily square).
Interspersion and Juxtaposition (IJI; Unit: Percentage, $0 < \text{IJI} \leq 100$)	IJI relates to the relationship that a cover class has with surrounding cover, quantifying the degree of interspersion between cover types at the patch-level. IJI reflects intermixing between adjacent land-cover, quantifying the unevenness of patch adjacency across the landscape. IJI approaches 0 when the corresponding cover is adjacent to only 1 other cover type. IJI = 100 when the cover type is equally adjacent to all other cover classes.
Cohesion (COHESION; Unitless, $0 \leq \text{COHESION} < 100$)	Measurement of structural connectivity of a single cover class across landscape. COHESION approaches 0 as the proportion of the landscape comprised of the focal class declines, becoming increasingly subdivided.Fragmentation of habitat will result in the landscape becoming increasingly disconnected, affecting ecological flows, and functional connectivity.

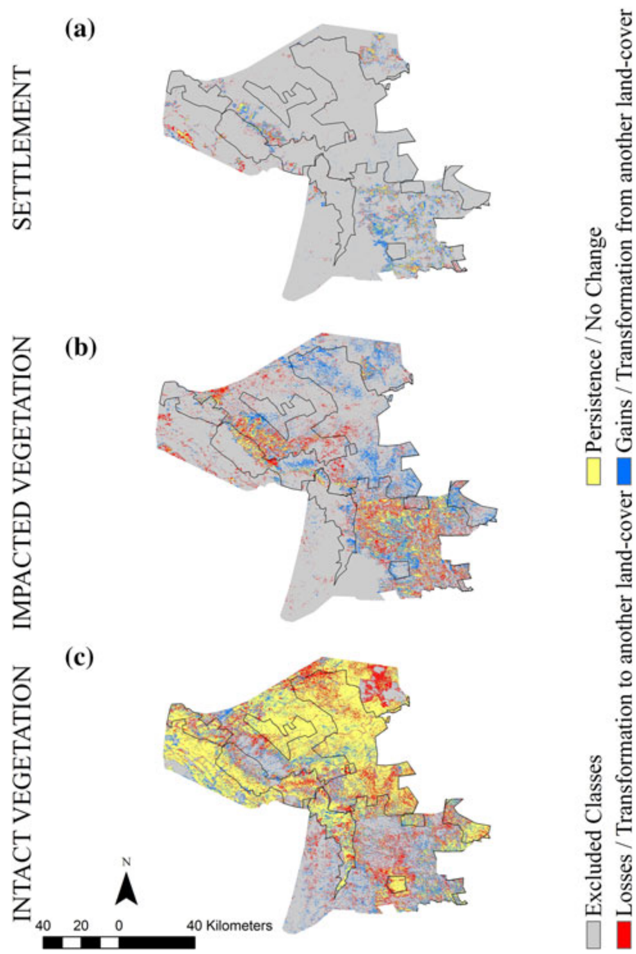

Fig. 5 Spatial patterns of Gains (G), Losses (L), and Persistence (P) (1993–2006) in **a** Settlement, **b** Impacted Vegetation, and **c** Intact Vegetation cover across the Kruger to Canyons Biosphere. *Note* The excluded cover classes are those cover types not included in the "selected cover class" categories of this article. However, they were defined for original classification maps

(Table 6, PLAND; $\sim$>390 km^2 vs $\sim$200 km^2), accounting for a 1.7 % ($\sim$>180 km^2) spatial increase since 1993 (Table 4, PLAND). Settlement expansion tracked transport routes (Fig. 5a); in particular, the arterial roads (established prior to 1993) toward KNP and major railway line across southern K2C. With the spatial expansion of Settlements, patch configuration was altered and patch shape complexity increased noticeably, an artifact of the ribbon development along roadways.

Settlement expansion was structurally cohesive (Table 6, $\text{COHESION}_{\text{gains}}$), compact areas of conversion, with "gain" patches likely to be associated with other "gain" patches [Table 6, low-Area Weighted Mean Shape Index (AWMSI = 3.8), Aggregation Index (AI) > 62 %]. This resulted in a more cohesive class across K2C (Table 4, COHESION). Spatially, Settlements not only expanded into the adjacent communal rangelands (Fig. 6e, f), transforming Impacted Vegetation ($\sim$150 km^2), Intact Vegetation cover ($\sim$>20 km^2), and exposed ground areas (<10 km^2) (Fig. 7), but also occurred within settlement areas, as the village space became more intensively utilized by the local settlement population. With this expansion, there was an increase in both the number of patches, and the average patch size across the BR, across all three zones [Tables 4, 5; No. of Patches (NP) and AWMPS].

However the landscape-scale trend of a more structurally cohesive class did not endure in the core zone. Here, the Settlement class remained maximally interspersed with surrounding cover types [Table 5, Interspersion and Juxtaposition approaching maximum (IJI = 99.66), lower COHESION], regardless that there were spatial gains made by this class here, i.e., larger patches, more abundant patches, affecting a greater proportion of this zone (Tables 5, 7; PLAND, AWMPS, NP). While the scarcity of Settlement explains this result to a degree, the nature of development in PAs may contribute further to an explanation, i.e., spatially distributed settlement developments, specifically to maintain tourist appreciation of an intact environment.

Across K2C, in contrast to the gains made by the class, Settlement losses (i.e., conversion *from* Settlement to another class) were substantially smaller sized, and more dispersed across the landscape (Table 6, $\text{COHESION}_{\text{losses}}$), occurring predominantly in the transition zone (Table 7,

Table 4 Class-level landscape metrics for selected land-cover classes in 1993 and 2006

Landscape metric	Intact Vegetation		Impacted Vegetation		Settlement	
	1993	2006	1993	2006	1993	2006
PLAND (%) of K2C	59.44	55.40	16.45	17.15	4.19	5.86
AWMPS (ha)	485,546	387,397	13,148	6,502	413	1,996
No. of patches	39,514	43,156	52,669	59,959	17,804	22,580
AWMSI	106.16	102.16	27.90	18.18	6.29	12.56
IJI (%)	57.28	54.31	62.83	69.27	99.45	96.12
AI (%)	91.36	90.31	78.69	77.46	74.96	77.82
COHESION	99.96	99.94	99.27	98.80	96.06	98.27

Table 5 Class-level metrics for selected land-cover classes for 1993 and 2006 specified by core, buffer, and transition zones in the Kruger to Canyons Biosphere Reserve where a) describes changes in Settlement, b) Impacted Vegetation, and c) Intact Vegetation

	Core		Buffer		Transition	
	1993	2006	1993	2006	1993	2006
Settlement						
PLAND (%) of K2C	0.02	0.06	0.29	0.35	3.90	5.45
AWMPS (ha)	31	44	43	216	436	2,058
No. of patches	225	1,199	3,009	3,877	14,709	17,695
AWMSI	2.28	2.0	2.93	4.32	6.48	12.85
IJI (%)	96.47	99.66	97.41	99.77	98.98	95.13
AI (%)	62.53	55.40	62.75	64.96	75.88	78.88
COHESION	83.24	77.97	87.45	91.97	96.33	98.40
Impacted Vegetation						
PLAND (%) of K2C	1.01	1.37	2.65	4.29	12.39	11.51
AWMPS (ha)	150	976	1,283	581	13,983	7,203
No. of patches	6,310	4,042	14,876	22,433	32,263	34,566
AWMSI	4.04	9.04	8.00	6.75	1.21	19.86
IJI (%)	6.80	26.10	30.51	27.79	73.22	85.19
AI (%)	70.01	79.34	73.83	72.23	80.34	79.05
COHESION	92.73	97.82	96.79	95.96	99.40	98.99
Intact Vegetation						
PLAND (%) of K2C	6.60	6.38	23.13	20.95	29.77	28.12
AWMPS (ha)	21,262	19,570	127,348	99,490	86,044	50,900
No. of patches	2,671	2,351	4,182	5,921	34,041	36,254
AWMSI	33.83	23.67	41.89	57.56	4.04	9.04
IJI (%)	9.79	28.76	39.46	25.62	65.54	69.00
AI (%)	90.03	91.97	94.73	92.04	88.76	88.39
COHESION	99.82	99.78	99.94	99.93	99.78	99.66

$PLAND_{losses}$). Spatially, Settlement losses occurred on the outskirts of village areas, where original settlements were recolonized by neighboring Impacted Vegetation of the communal rangelands, or converted for agriculture, mostly for subsistence purposes (Fig. 7). Losses also occurred within villages (Fig. 5a)—where settlement cover was lost to the creation of garden-plots associated with household livelihoods, with loss patches more regularly shaped, square units compared with areas of Settlement expansion/persistence (Table 7, $AWMSI_{losses}$).

Impacted Vegetation

As with the Settlement class, the Impacted Vegetation cover class was associated with areas of intensive human activity (Fig. 5b), with 67 % of the total 2006 Impacted Vegetation cover found in the transition zone of the BR (Table 5, PLAND; Fig. 6e, f).

By 2006, Impacted Vegetation increased to 17.2 % of K2C (Table 4, PLAND, net increase >120 km^2) having expanded into the neighboring Intact Vegetation areas (Figs. 5, 7), with gain patches affecting 9.5 % (~1,060 km^2) of K2C (Table 6, $PLAND_{gains}$). This expansion was accompanied by declines in patch size and structural cohesion across K2C, while patch shape irregularity increased (Table 4, AWMPS, COHESION, AWMSI), with greater edge sharing and patch adjacencies with other land-cover types (Table 4, increased IJI in 2006).

Most gains occurred across the transition zone (Table 7, $PLAND_{gains}$, Fig. 6e, f) and to a lesser degree in the buffer zone (Table 7, $PLAND_{gains}$; Fig. 6c, d, e.g., western K2C buffer zone), as compact patches of conversion (Table 7, AWMSI < 5) associated with pre-existing patches of cover across all zones (Table 6, IJI > 70 %; Table 7, IJI > 65 %).

Although the net change within this cover class was spatial increase across K2C (Table 4), Impacted Vegetation losses occurred over 8.8 % of K2C (Table 6, PLAND; ~940 km^2) with a substantial portion of Impacted Vegetation in 1993 having been transformed to Settlement by 2006 (>150 km^2, Fig. 7). As villages expanded, they become more crowded, and environmental resource needs increased commensurately. As a result, losses in Impacted Vegetation cover were structurally cohesive areas of transformation (Table 7, PLAND, COHESION;

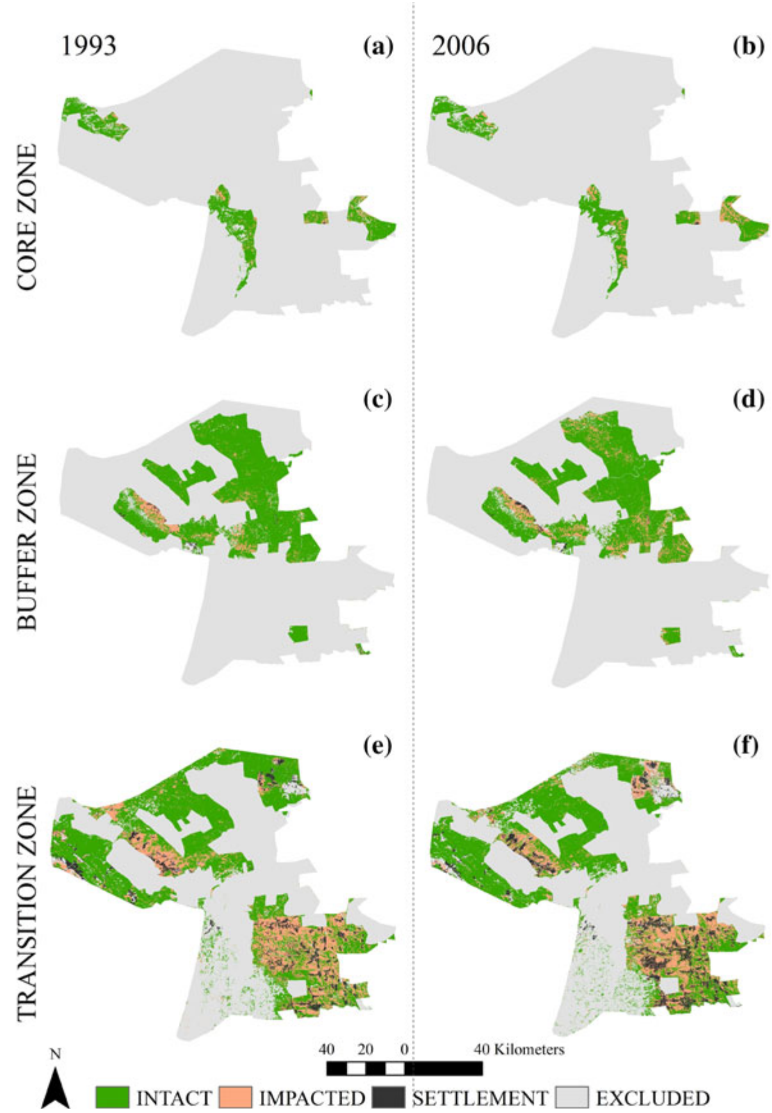

Fig. 6 Spatial distribution of Intact Vegetation, Impacted Vegetation, and Settlement classes in the Kruger to Canyons Biosphere Reserve (K2C), relative to spatial zonation for 1993 and 2006. *Inset* **a**, **b** (Internal) Core zone, *Inset* **c**, **d** Buffer zone, and *Inset* **e**, **f** Transition zone

Table 6 Class-level landscape metrics for Gains (G; conversion to selected cover *from* another cover type), Losses (L; conversion from selected cover to another cover type), and Persistence (P; spatial extent of selected cover had no change) of selected land-cover classes in the Kruger to Canyons Biosphere Reserve (K2C) between 1993 and 2006

Landscape metric	Intact Vegetation			Impacted Vegetation			Settlement		
	G	L	P	G	L	P	G	L	P
PLAND (%) of K2C	10.23	14.23	45.21	9.52	8.80	7.64	3.62	1.26	2.24
AWMPS (ha)	33	532	303,077	130	76	323	80	8	148
No. of patches	127,561	129,366	51053	77,972	72,825	38,410	30,466	20,412	9,425
AWMSI	3.12	6.58	119.94	4.48	3.81	6.35	3.82	2.05	5.07
IJI (%)	54.75	49.23	99.40	75.39	72.06	99.80	65.96	82.67	95.04
AI (%)	61.21	67.79	89.41	63.60	62.32	72.21	62.99	49.19	73.45
COHESION	87.85	95.12	99.95	92.04	90.31	95.71	90.56	76.06	94.90

Figs. 6e–f); located predominantly at the peripheries of the existing settlement areas, and to a lesser degree, alongside the commercial agricultural areas in central K2C (Fig. 5b). Thus, Impacted Vegetation losses converged in the transition zone (Tables 5, 7, PLAND), with losses here exceeding gains made by this class (Table 7, PLAND),

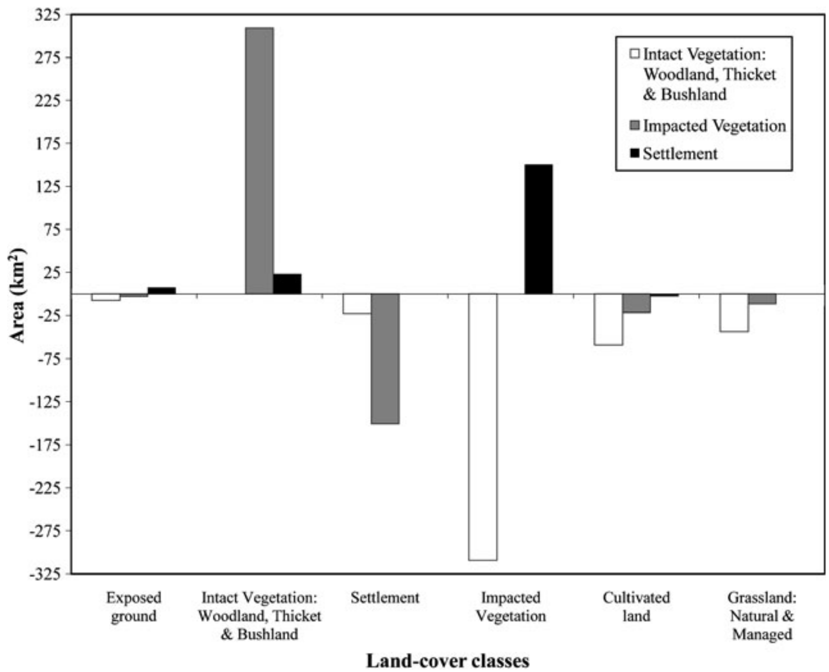

Fig. 7 Contributors to changes (expansion and reduction) in spatial extent of selected land-cover classes detailed in this study (Intact Vegetation, Impacted Vegetation, and Settlement) across the Kruger to Canyons Biosphere Reserve (K2C) between 1993 and 2006. Additional classes beyond these that have been displayed on the *x*-axis (Exposed ground, Cultivated land and Natural, and Managed grassland) are present in K2C but not the focus of this study (described in Coetzer et al., 2010 in detail). Interpretation note: Positive values on *y*-axis indicate *expansion into-*, and *transformation of*—the land-cover classes presented on the *x*-axis by Intact-/Impacted/Settlement cover, (i.e., Intact-/Impacted/Settlement cover enlarging in extent). *Negative values* indicate *loss of-* the Intact-/Impacted/Settlement cover through *conversion to-* the land-cover classes displayed on the *x*-axis (i.e., reduction in extent by Intact-/Impacted/Settlement cover class due to transformation to another cover-type)

resulting in an overall decline in the Impacted Vegetation cover in this zone (Table 5).

In contrast to the transition zone, and aligned with the trends observed at the landscape scale, Impacted Vegetation cover increased in both core and buffer zones, with the majority of these landscape scale increases driven by those of the buffer zone (Table 5, PLAND). In the buffer zone, patches became on average smaller sized and more geometrically compact over time (Table 5, AWMPS, NP, AWMSI), with the spatially contiguous areas of Impacted Vegetation cover most frequently located outside the protection afforded by the privately-owned reserves of this zone (Fig. 6e, f). In the core zone, the increased IJI, indicated greater interspersion and intermixing with surrounding cover across the zone (Table 5). This suggests that the Impacted Vegetation class was less confined to specific localities here, i.e., only on the peripheries of each PA, and rather more evenly spatially distributed across the zone (Fig. 6a, b), becoming more prevalent and structurally contiguous in the core PAs with time.

Intact Vegetation

The Intact Vegetation cover class dominates across central K2C, and along the escarpment area in the west (Fig. 5c), where it exists as a physically connected expanse across all three zones (Tables 4, 5, high COHESION). In 2006, it accounted for 55 % (>6 100 km^2) of all available land in K2C, a >4 % (~>400 km^2) reduction in its 1993 coverage (Table 4, PLAND), with smaller, more numerous patches located across K2C. Patch shape remained irregular in 2006, with complex shape geometry associated with greater edge sharing with surrounding cover (Table 4, AWMSI > 100). This combined with a more fragmented Intact Vegetation expanse may have important consequences for the future transformation of this class.

Given the protection status in the core, and to a lesser degree in the buffer zone, the Intact Vegetation class constitutes the largest percentage cover contribution across all zones (Fig. 6; Table 5). However, spatially, the majority of this class is located in the transition zone in both 1993 and 2006 (Table 5, PLAND; >50 % of the total extent in both years). This is an artifact of the larger spatial extent of the transition zone relative to the core and buffer, rather than a specific management approach here. In the core and buffer zones, Intact Vegetation had greater structural cohesion and spatial aggregation (Table 5, COHESION, AI), in comparison with reduced fragmentation across the class in these areas.

The total spatial losses of Intact Vegetation were physically connected across the original Intact Vegetation

Table 7 Class-level metrics for Gains (G; conversion *to* selected cover from another cover type), Losses (L; conversion *from* selected cover to another cover type), and Persistence (P; areas of no change in spatial extent): a) Settlement, b) Impacted Vegetation, c) Intact Vegetation land-cover (1993–2006) as specified by spatial zonation

	Core			Buffer			Transition		
	G	L	P	G	L	P	G	L	P
Settlement									
PLAND (%) of K2C	0.057	0.008	0.007	0.26	0.19	0.09	3.22	1.67	2.23
AWMPS (ha)	11	1	58	26	12	42	103	25	177
No. of patches	1,422	262	37	4,952	3,828	900	29,468	24,857	11,619
AWMSI	2.08	1.38	3.63	2.73	2.11	3.30	4.21	2.59	5.42
IJI (%)	45.79	89.38	70.50	76.22	86.21	97.95	63.15	75.97	97.17
AI (%)	53.28	47.58	82.41	57.50	55.47	70.69	67.51	59.32	76.22
COHESION	78.72	63.34	95.46	85	79.63	91.40	92.46	84.73	95.73
Impacted Vegetation									
PLAND (%) of K2C	0.93	0.57	0.44	3.19	1.56	1.09	5.47	6.36	6.04
AWMPS (ha)	94	13	69	97	77	277	112	71	283
No. of patches	6,400	8,259	4,010	29,673	19,673	10,056	57,237	55,824	32,314
AWMSI	4.43	2.34	3.24	4.39	3.10	5.17	4.34	4.14	6.51
IJI (%)	67.27	83.48	95.26	77.35	77.96	99.99	68.94	63.94	99.57
AI (%)	72.17	60.70	70.95	68.07	63.67	72.64	65.74	67.01	75.62
COHESION	93.93	83.73	91.27	92.91	88.55	94.83	92.26	92.20	96.33
Intact Vegetation									
PLAND (%) of K2C	1.28	1.5	5.1	2.19	4.37	18.76	6.76	8.36	21.37
AWMPS (ha)	20	65	14,061	16	162	69,019	39	682	38,712
No. of patches	14,877	12,184	4,654	31,566	38,109	10,640	82,812	80,312	36,992
AWMSI	2.76	3.76	33.96	2.6	4.98	55.79	3.29	7.07	37.09
IJI (%)	44.48	52.33	98.69	53.0	36.36	93.67	58.05	57.31	99.99
AI (%)	62.73	69.28	87.98	58.70	67.89	91.09	61.57	67.34	88.06
COHESION	86.82	92	99.74	84.51	93.66	99.9	88.58	95.45	99.72

expanse (Table 6, high COHESION), affecting >14.23 % (~1,570 km^2; Table 6, $PLAND_{losses}$) of K2C, with net transformation (i.e., total gains minus total losses), occurring to Impacted Vegetation (>300 km^2), Cultivated land (~60 km^2), Settlement (>20 km^2), and Grassland cover (managed and natural; >40 km^2) (Fig. 7). Individual patches of loss were compact in shape (Table 6, low $AWMSI_{losses}$), associated with other areas also undergoing conversion (Table 6, lower IJI = 49.23). This too has consequences for future conversion of this class, suggesting a trend of cohesive transformation, rather than isolated occurrences across the subregion.

While conversion of Intact Vegetation was associated with all three spatial zones across K2C to some degree, the pattern of loss is not spatially consistent throughout (Table 5; Fig. 6). The majority of the losses occurred in the buffer and transition zones, predominantly areas lacking structured environmental protection. With reduced protection, losses were larger in size (Table 7; $AWMPS_{losses}$), resulting in more numerous, smaller-sized patches of Intact Vegetation across each zone in 2006 (Table 5, NP, AWMPS).

Given the transformation of Intact Vegetation beyond the densely-populated settlement expanses of the transition zone (Fig. 5c), the losses of Intact Vegetation extended into the formally delineated conservation areas. Conversion of Intact Vegetation intruded into the privately- and community-owned PAs of central and eastern K2C (Fig. 5c); directly both core and buffer zone PAs (Fig. 1a). For the PAs within Bushbuckridge Municipality, e.g., Bushbuckridge Nature Reserve, Manyeleti (buffer zone) and Andover Reserves (core zone), conversion to Impacted Vegetation has been substantial on the PA boundaries, and converged on the center of the reserve (Fig. 6c–f). While the “hard fences” of these PAs (with the exception of state-owned PAs under weak management) will prevent wide-scale losses and the total transformation of Intact Vegetation here, land-cover change of the surrounding area may have potential long-term consequences for a landscape of ‘island’ PAs within a transformed matrix.

However, the transformation of Intact Vegetation was not consistently unidirectional, although the net change trend was one of loss (Fig. 7; Table 4). Spatially, gains made by the Intact Vegetation cover class predominated within central K2C, and the communal rangeland areas of the transition zone (Fig. 5c; Table 7, $PLAND_{gains}$) affecting 10.2 % (~1,120 km^2) of K2C (Table 6, $PLAND_{gains}$). These gains were compact areas (Table 6, low AWMSI), small in size (AWMPS = 33 ha), frequently associated with other (Gain) areas (IJI = 54.75). Based on field observations, this is most likely indicative of resource harvesting and the densification of woody vegetation cover (Fig. 2a–c) spatially disaggregated across K2C (Table 6).

Discussion

General Patterns of Change

Spatially, changes observed in the priority classes radiated outward from the dense settlement expanse and areas of intensive human activity of the transition zone (Fig. 5). Settlements have grown multi-directionally and village boundaries have converged. The changes observed in Impacted Vegetation and Intact Vegetation classes concentrate around the Settlement areas, becoming less extensive with distance (Fig. 5b, c), demonstrating the landscape-scale effect of disturbance gradients around settlements (Shackleton and others 1994; Fisher and others 2012).

The "advancing urbanization" and construction of new houses in rural landscapes typically lead to the rapid further transformation of surrounding natural areas (Kjelland and others 2007), with the subsequent degradation of original environmental values. While village population sizes have near stabilized or shown only slight increases between 1993 and 2006 in the subregion (Fisher and others 2012), there have been notable changes in household sizes. Within K2C, the trend is toward smaller households, with significant decreases occurring during 1997–2003 (Madhavan and Schatz 2007). This, in conjunction with an even slightly increasing regional population, will result in a greater transformation to the Settlement class over time than explained by total population growth alone.

The spatial expansion of the Settlements encroached into adjacent Impacted Vegetation (Figs. 5, 6, 7). Impacted Vegetation shifted spatially in response, expanding into areas of original Intact Vegetation cover (Fig. 5b–c), as the Settlement's zone of influence and associated environmental utilization extended into the surrounding landscape. The extensive Intact Vegetation has increasingly fragmented across K2C as a result, eroded by transformation to the other land-cover classes, particularly on the peripheries of its original 1993 distribution (Fig. 5c; Table 4).

The distribution of these changes bears little regard for the theoretical zonation and limitations on land-cover change that typify the implementation of a successful "Biosphere Reserve," with land-cover changes occurring beyond transition zone, into core and buffer zones (Fig. 6). This has consequences for K2C's compliance with the idealized model expected of BRs (e.g., "Land-cover changes: Issues for K2C's 'Biosphere Reserve' status," also Coetzer and others 2013). A diverging trajectory between urbanization, conversion, and homogenization across the matrix landscape of transition and unprotected buffer zone versus the "pure" conservation of the fenced core and buffer zone PAs, with no graduated scale of land-use intensity, may predict the dissolution of the BR model.

Changes in Impacted Vegetation: Issues Around Sustainability

Our results indicate that as settlement areas enlarged and became more crowded, so too did their footprint of resource utilization (Fig. 5b; Table 4), and the expansion of Impacted Vegetation consolidated the communal-use rangelands of neighboring villages. Rangeland boundaries have become indistinct, and spatially (as opposed to functionally) sprawling rangelands are not restricted to specific village units. This amalgamation has serious implications for the long-term sustainability of these existing communal rangelands as resource harvesting areas, especially if growing settlement expanses rely on consolidated rangelands offering lower-quality rangeland products (e.g., Higgins and others 1999; Matsika and others 2013a, b; Wessels and others 2013).

Previous studies in the subregion have indicated that as fuelwood resources are depleted around villages, residents travel greater distances to gather the required wood, increasingly using wheelbarrows and vehicles to transport the harvest home (Giannecchini and others 2007). This has potentially severe consequences for the future exploitation of the Intact Vegetation located outside PAs in the transition and buffer zone (Fig. 6c–f); a cycle of more efficient transport methods resulting in greater harvest loads, with vehicle passage improving access to previously unavailable extraction areas.

Management Options for K2C: Preventing Further Losses of Intact Vegetation

Losses of Intact Vegetation occurred primarily through cohesive areas of transformation to Impacted Vegetation cover (Fig. 5b, c), with the majority of the conversion associated with an existing (persisting) patch of Impacted Vegetation (Table 6). This suggests that large areas of transformation to Impacted Vegetation require a "kernel" of existing Impacted Vegetation cover to initiate conversion [i.e., the initial (vehicle) foray for fuelwood harvesting].

This has direct consequences for the scheduling of management action in K2C, as unchecked, these nuclei of Impacted Vegetation may set the trajectory for rapid and large-scale transformation of Intact habitat. Effective management strategies should prioritize these small nuclei of future transformation, e.g., areas in central and eastern K2C buffer zones (Figs. 5b, c, 6c, d), and prevent their establishment and/or enlargement, with stricter controls on access to intact and important biodiversity sites required. Failure to do this, and the potential for extensive fragmentation of the priority Intact Vegetation cover may escalate rapidly, with knock-on effects for the landscape connectivity of the class across K2C in the future.

Management Options: Maintaining Connectivity of Intact Vegetation

From 1993 to 2006, Intact Vegetation maintained high-structural connectivity across K2C (Table 4), across all zones (Table 5); its persistence and spatial cohesion attributed to the conservation afforded by existing PAs in the central part of the BR (Fig. 1a), and the historical inaccessibility of the mountainous western escarpment region.

While an overall loss of >4 % across the landscape of Intact cover may appear minor, landscape effects will mask dramatic changes at finer scales, as in the case of Matsika and others 2013a, b. If combined into a single area, it becomes spatially substantial (>400 km^2) beyond the size of many existing PAs in K2C, e.g., Andover, Bushbuckridge Nature Reserves, Manyeleti.

Similarly, the spatial extent of these losses of Intact Vegetation is important, as their distribution directly influences potential linkages between existing PAs. Intact Vegetation suffered both "block losses" (large areas of conversion, located predominantly in transition zone) and "salt and pepper" losses (smaller, dispersed areas of conversion located predominantly in the buffer and core zones) across K2C (Table 7; Fig. 5c). Block losses denote an immediate concern having resulted in an increasingly fragmented macro-landscape. Yet the smaller "salt and pepper" losses may act as a precursor to the erosion of larger areas, affecting management decisions around scheduling future conservation action. Once large-scale habitat is lost, the matrix frequently becomes set on a trajectory of further land-cover conversion and associated habitat degradation (Brady and others 2009).

For the southern part of K2C, this management action targeting smaller areas of transformation may offer great value. The losses of intact habitat around the Bushbuckridge Nature Reserve (buffer zone PA) may have severe consequences for the conservation proposals to functionally connect the core PAs: KNP to the smaller PAs across Mpumalanga Province (e.g., Motlatse Canyon National Park (formally *Blyde River Canyon National Park*, Fig. 1a). At present, Bushbuckridge Nature Reserve may represent a sufficiently large refuge for biodiversity in this intensively utilized transition zone. It offers longer-term functional habitat opportunities here, facilitating "stepping-stone" dispersal across this largely inhospitable area.

However, the process of fragmentation entails more than just habitat loss (Fahrig 2003), with patch configuration also important (McGarigal and Cushman 2002; Fahrig 2003; Mortelliti and others 2010). With loss of Intact Vegetation, patch shape geometry became more complex and spatially dispersed across the class (Table 4), with greater edge sharing with other classes, e.g., Impacted Vegetation (Fig. 5b, c). This increases the potential for enhanced edge effects with adjacent cover (e.g., invasion of pioneer or exotic species from the rangelands). Given that edges are considered a driving force behind fragmentation and subsequent environmental degradation (Murcia 1995; Harrison and Bruna 1999; Fletcher 2005), the potential for edge sharing of the Intact Vegetation with other classes, and already transformed (loss) patches (Table 6) may initiate further transformation and fragmentation of Intact Vegetation cover. Hence, the type and distribution of change revealed by the GLP maps may underestimate the ecological effects in the landscape; potential ecological ramifications that may extend beyond the exact locations of the losses in- or gains of- a particular land-cover type.

However, ensuring the preservation of habitat quality of smaller habitat fragments across K2C may also have conservation merit (Prugh and others 2008; Brady and others 2009). The gains of Intact Vegetation within the original Impacted Vegetation expanse (Table 6; Fig. 5c), while small and isolated act to "soften" (*sensu* Franklin 1993) the structure of the Impacted Vegetation here and facilitate movement across the transition zone. They contribute to increasing heterogeneity of sparsely vegetated rangeland areas, and in terms of maximizing conservation rewards, the densely vegetated, bush encroached areas may also offer rehabilitation possibilities that exceed those of more degraded, or sparsely vegetated patches.

Field observations indicate that these gains are likely incidents of bush encroachment [i.e., spectrally similar to that of the Intact Thicket cover class (Table 1; Fig. 2c)]; conversion from the original sparsely wooded, exposed vegetation of the Impacted Vegetation class to a more densely wooded cover as a result of human utilization of the environment, regardless of increased electrification in the subregion [i.e., major cost-savings in fuelwood use vs electricity tariffs (Madubansi and Shackleton 2007), irrespective of socio-economic context of individual villages (Matsika and others 2013a, b)].

Land-Cover Changes: Issues for K2C's "Biosphere Reserve" Status

Maintaining Ecological Integrity and Existing Conservation Values

The zonation within BRs (core:buffer:transition) is intended to protect the core biodiversity areas and associated Intact Vegetation from habitat conversion and escalating fragmentation associated with land-cover change. Yet in many developing countries, the land-cover change in the transition zone places the conservation objectives of the core zone under considerable pressure (Ma and others 2009).

For K2C, spatial limitations on available land area undermine the effectiveness of the BR conservation model in the long term. The expanding settlement areas and their resource requirements (e.g., expanding Impacted Vegetation), local economic instability (entrenched poverty and few opportunities for upliftment), and the land-use requirements of the economic sectors (Coetzer and others 2010) may severely compromise the ability of non-conservation land uses to remain restricted to the transition zone. The BR model here has been applied to a socio-ecological landscape, one with a pre-existing spatial distribution of existing land uses, dynamic in nature, and unlikely to be governed by an "imaginary" boundary restricting resource use.

Should the trends outlined continue, the potential for transformation up to PA fences is significant. This raises serious concern about the ability of the matrix landscape, specifically the communal rangeland areas of the transition zone, to continue to deliver services to the resource-dependent rural settlements and the core and buffer PAs located within, and adjacent to this transition zone (Fig. 1). A transformed matrix of homogenous vegetation cover, fueled by the cycle of more extensive and intensive use of rangelands by the (increasingly resource deprived) expanding settlement areas (Matsika and others 2013a, b; Wessels and others 2013), will not offer the same adaptation opportunities for nearby PAs as a heterogeneous landscape (i.e., for connectivity purposes under global change scenarios, and for ecosystem services). Similarly, neither will homogenous rangeland areas retain the ability to provide adaptation options for rural settlement residents in times of stress.

Thus, based on the transformation patterns we present here, immediate management priorities must target vulnerable PAs, and the Intact Vegetation areas closest to potential "transformation zones". Possible management interventions aimed at controlling vehicle access in specific areas (i.e., community conservation areas in Bushbuckridge) or specific buffer zones around individual PAs may offer value for these sites.

Conclusions for Biosphere Reserve Management

For the longer term, management priorities should re-evaluate the validity of the MAB model for this South African scenario. Priority should be to reassess the spatial arrangement of the BR zonation. Should the zonation not translate in practice, with buffer and transition zones not fulfilling buffering and/or connectivity functionality, or restricting resource use, a different and/or additional management strategy may be required beyond only relying on theoretical zones. This, however, is not only restricted to the South African scenario, but has wider-scale international applicability: relying on the merits of zonation that does not fulfill the BR prescription is an inefficient exercise in general.

The re-evaluation of the zonation will need to be an adaptive management initiative that is repeated at certain time intervals, so as to ensure forward looking, proactive management decisions, e.g., see South African National Parks (SANParks) strategic adaptive management approach for resource conservation (Biggs and Rogers, 2003). To be effective, it will need to utilize historical land-cover data (similar to that presented here), incorporate K2C's conservation priorities and the socio-economic objectives of the South African Government in the redesign of the buffer and transition zones.

The land-cover changes we have highlighted here are unlikely to have been driven by the designation of the BR in the subregion, given that the site was only listed in 2001. Similarly, the current economic developments planned/successfully piloted thus far by the K2C management authorities, or those associated with the national PAs (i.e., for details Coetzer and others 2013), are unlikely to fuel uncontrolled land-cover conversion in their current form, given that at this point they predominantly remain site-specific localities. In many instances, they have provided resident communities with previous inaccessible opportunities, e.g., employing members for monitoring the function of the BR (*Environmental Monitor Project*), or resource harvesting projects in KNP (e.g., Swemmer and others 2011; Scheepers and others 2011).

Instead, the land-cover change we have outlined here has occurred *regardless* of the subregion's listing with MAB, rather than as a result. This is unlikely to be specific to only a South African scenario given that new sites are listed with MAB each year (Coetzer and others 2013). However, it is this that may compromise the longevity or successes of the BR model long term (specifically at the level of zone delineation), if it is overlooked.

K2C is different to most other BRs, in that both core and buffer zones are largely formally-fenced conservation areas, and likely to remain in the landscape long term, and their complete transformation is improbable. Where BRs located elsewhere have had reduced implementation successes, this has been due to inter alia: (1) improperly engaging with BR stakeholders in the BR, or (2) underestimating the ecological consequences of implementing development/economic initiatives (reviewed in Coetzer and others 2013).

With reference to the first, K2C has made noteworthy achievements in this regard, and was nominated for a 2013 UNESCO award (formally: *Michel Batisse Award for Biosphere Reserve Management*, www.unesco.org). With reference to the second, this study and future studies that follow on from this one, i.e., tracking land-cover change as a trend, rather than a two (time)-point analysis, is critical, providing a base-line for zone delineation decisions, provided zone delineation is adaptively managed.

Such an adaptive management remote-sensing approach to zone management has the potential for upscaling to other BR sites internationally, particularly those listed in areas where the landscape also has substantial utilization pressures. This is not only restricted to landscapes in which poverty or population pressures fuel environmental utilization, but also to those BRs listed in affluent landscapes; i.e., with the notion that land-cover change is not only caused by the "poor". Few BRs globally have had strategic re-evaluation of zonation documented in the literature (Coetzer et al., 2013; Ma and others 2009 presents an exception), regardless that the codevelopment:conservation aspirations are likely to impact upon and be affected by the underlying socio-ecological land-use mosaic.

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Chapter 5

Updating the time-series (1993 – 2012): Land-cover change and Intensity Analysis of change in a multi-use conservation landscape of South Africa

Updating the time-series (1993 – 2012): Land-cover change and Intensity Analysis of change in a multi-use conservation landscape of South Africa

5.1 Abstract

The reality of contemporary conservation has become environmental management within the context of human-modified landscapes, with matrix land-cover influencing the longer-term sustainability of existing protected areas. Biosphere Reserves (BRs), as multi-use conservation landscapes, exemplify the landscape mosaic approach to environmental decision-making. The Kruger to Canyons Biosphere Reserve (K2C) of north-eastern South Africa is the focus of this study. K2C is a spatially structured socio-ecological system, with important economic sectors and world-renowned conservation areas proximal to extensive, mostly rural, human populations engaged in informal livelihood practices. These vast settlement expanses are artefacts of the former political system in South Africa, and have been prioritised for urgent socio-economic upliftment. In this study, time-series remotely-sensed data (1993 – 2006 – 2012) is used to monitor land-cover change across the Biosphere Reserve subregion, updating previous land-cover research in K2C with the addition of 2012 Landsat imagery. The focus is on changes in the scale of land-cover change (spatial extent, rate and intensity of change) across the analysis period, applying the Intensity Analysis approach of Aldwaik and Pontius (2012), and simulating future changes to 2018. Time-series analysis has revealed new insight into the process of land-cover change across K2C, information that would have been overlooked had a two-point observation of change been retained. The increased rate of change in the recent observation period (2.3% versus 5.7%) poses challenges for landscape management, with future predictions of escalating transformation likely to undermine BR sustainability. Herein, the thresholds for scheduling action have been identified, allowing for timeous detection while practical options for intervention remain.

Keywords: Intensity Analysis; Kruger to Canyons Biosphere Reserve; landscape mosaic; natural resource management; socio-ecological system; time-series analysis

5.2 Introduction

Globally, there is increasing awareness that traditional protected areas (PAs) are becoming ecologically isolated (Svancara et al., 2009), and dependant on surrounding matrix land-cover for long-term sustainability (Prugh et al., 2008; Frankin and Lindenmayer, 2009). The reality of contemporary conservation has become environmental management within the context of human-modified landscapes (Western, 2001; Chazdon et al., 2009), with conservation approaches having to consider the benefits of monitoring the matrix landscape beyond the 'protected area' (Prugh et al., 2008). Studies have indicated the importance of the total extent of natural vegetation remaining across the landscape mosaic for ecosystem resilience and the precipitation of regime shifts (Pardini et al., 2010), with the quantity and quality of intact vegetation cover in the matrix affecting spatial linkages, the potential for ecosystem homogenisation and the loss of historical disturbance regimes across the region (reviewed in Western, 2001; Fischer and Lindenmayer, 2007; Gardner et al., 2009). Thus, the value of the matrix, and the value of formalising the notion of a 'multi-use conservation landscape' for biodiversity management cannot be underestimated.

This has fuelled the need to understand the landscape context of PAs at multiple scales of measurement (Wiens et al., 2008; 2009), tracking land-cover change across administrative boundaries and management objectives (Wiens et al., 2009; Svancara et al., 2009). Such broad-scale monitoring inherently acknowledges the landscape mosaic as a spatially structured socio-ecological system (Cummings et al., 2011), and indicates the complexities of the relationships between conservation and socio-economic factors in the wider landscape (de Fries et al., 2007). In this regard, Biosphere Reserves (BR) are uniquely positioned to take advantage of this systems thinking approach to conservation (e.g. Nguyen and Bosch, 2012), explicitly including human-resource use, and the matrix land-cover within the landscape mosaic, as a central tenet of their management strategy (UNESCO, 1996; Coetzer et al., 2013a).

The Kruger to Canyons Biosphere Reserve (K2C) of north-eastern South Africa is the focus of this study. Strong relationships between the local human population and the environment, across all scales, shape the distribution and integrity of biodiversity features in this landscape (Giannecchini et al., 2007). While the extent of these human-environment interactions (e.g.

Shackleton and Shackleton, 2002; Dovie et al., 2005) has helped ensure the subregion its 'Biosphere Reserve' status, these interactions in the future may severely compromise the successful implementation of the BR model here long-term (Coetzer et al., 2013b), with trade-offs between human use and longer term conservation likely. Thus for this BR, the value of spatio-temporal analysis of landscape change is considerable, with the potential to reveal unfavourable change trajectories while options for intervention remain financially, and spatially, viable.

Conservation interventions are typically constrained by available resources, and as such, conservation approaches have converged on those strategies that maximise return on investment (Polasky, 2008). Applying this methodology to change detection requires that changes that are temporary or spurious are not prioritised within the decision-making process (i.e. Aldwaik and Pontius, 2012), with effective monitoring translating into practical application, i.e. informing the scheduling urgency for management action.

However, this raises the specific challenges of *i*) identifying what constitutes an significant / important spatial change for any particular landscape over time, and *ii*) determining a critical spatial and temporal threshold of land-cover change for action.

In this regard, I argue that the multi-time point approach proposed by Aldwaik and Pontius (2012), hereafter *Intensity Analysis*, may offer considerable interpretative opportunities. The approach simultaneously compares land-cover changes across multiple time intervals, identifying stationarity in change trends, as well as providing quantitative insights into the variation in spatial distribution, the rate and intensity of land-cover change over the entire analysis period.

In this study Intensity Analysis has been applied to land-cover change trends in K2C. The method has been applied successfully on most continents (e.g. Aldwaik and Pontius, 2012; Huang et al., 2013; Kabanza et al., 2013; Pontius et al., 2013), providing the understanding necessary to link the spatial pattern of land-cover change to the drivers of change, and thus, appropriate management strategy. Yet, I am not aware of any previous Southern African application of Aldwaik and Pontius's multi-time point approach at this time.

Medium-resolution Landsat imagery has been utilised to characterise regional land-cover and the change dynamics from 1993, 2006 and 2012 across K2C. The original change detection period has been updated with the addition of the 2012 imagery, and specifically considers changes in the scale (spatial extent, rate and intensity of change) of landscape transformation across the time series. These data have been utilised to infer predictions of future landscape change for K2C until 2018 (+ 6 years) through a Cellular Automata-Markov modelling approach, with a consideration for long-term sustainability for this BR, and provide a potential critical threshold of change for the loss of intact habitat in this landscape.
As with the previous analysis (Coetzer et al., 2010, 2013b), the focus is on consolidated land-cover classes that are proxies for the dual management objectives of all BRs globally: development (Settlement) and human-driven resource use (Impacted Vegetation, e.g. communal rangelands) *versus* environmental protection (Intact Vegetation) across K2C.

5.3 Methods

5.3.1 Study site

The Kruger to Canyons Biosphere Reserve (K2C, http://www.kruger2canyons.org/, Fig. 1) was registered with UNESCO's Man and the Biosphere Programme (MAB) in 2001, and is currently one of five Biosphere Reserves (BRs) located in South Africa. At approximately 2.5 million hectares in extent, K2C is the second largest of these South African BRs, as well as one of the largest BRs listed globally.

Located in north-eastern South Africa (centred on 24.428°S, 30.827°E; Fig. 1b,c), K2C traverses two national administrative districts, the Limpopo and Mpumalanga provinces, each with associated separate provincial conservation agencies. It includes three distinctive natural biogeographic regions (*biomes*): Savanna, specifically dry savannah woodland, which predominates across the subregion, and Afromontane Forest and Grassland areas found in the higher altitudes of the escarpment (Mucina and Rutherford 2006). Altitude increases along an east-west gradient to the escarpment (200 – 2059 m above sea level), with reciprocal increases in average rainfall (400 mm: central low-lying areas – 3000 mm: escarpment), with most rainfall falling in the summer months (Oct. – Feb.). Average summer temperatures exceed 27°C. (http://www.kruger2canyons.org/).

The BR is a ‘multi-use conservation landscape’, with the BR designation applied to a pre-existing land-use mosaic (Coetzer et al., 2013b). A vast array of world renowned conservation areas, including the central portion of the Kruger National Park (KNP, full extent ~ 2 million hectares), are spatially juxtaposed against nationally important mining, commercial agriculture and forestry sectors (Fig. 1a), and a settlement expanse that continues to expand.
K2C’s resident population size is in excess of 1.5 million people, most of whom reside in rural communities, where despite the semi-arid environment population densities can exceed 300 people/km^2 (Pollard et al., 2003).

These rural communities, specifically those located within Bushbuckridge and Maruleng Municipalities (Fig. 1a) have been afforded ‘Presidential Poverty Node’ status, and identified as national priorities for urgent economic and development interventions (RSA, 2000). The high population densities, entrenched poverty, unemployment and historical marginalisation have resulted in locally harvested natural resources as essential to ensuring livelihoods (Shackleton and Shackleton, 2002; Dovie et al., 2005; Twine, 2005, Madubansi and Shackleton, 2007). This fundamental reliance on communal rangeland products has not abated following infrastructure improvements in these rural areas, and for example, even with increased electrification, recent studies are indicating that fuelwood harvesting may soon be reaching unsustainable levels (Fisher et al., 2012; Matsika et al., 2013; Wessels et al., 2013). This raises some concerns about the local environment’s ability to continue to deliver services to the resource-dependent settlement areas long-term (Coetzer et al., 2013b), with repercussions for the resilience of this socio-ecological system.

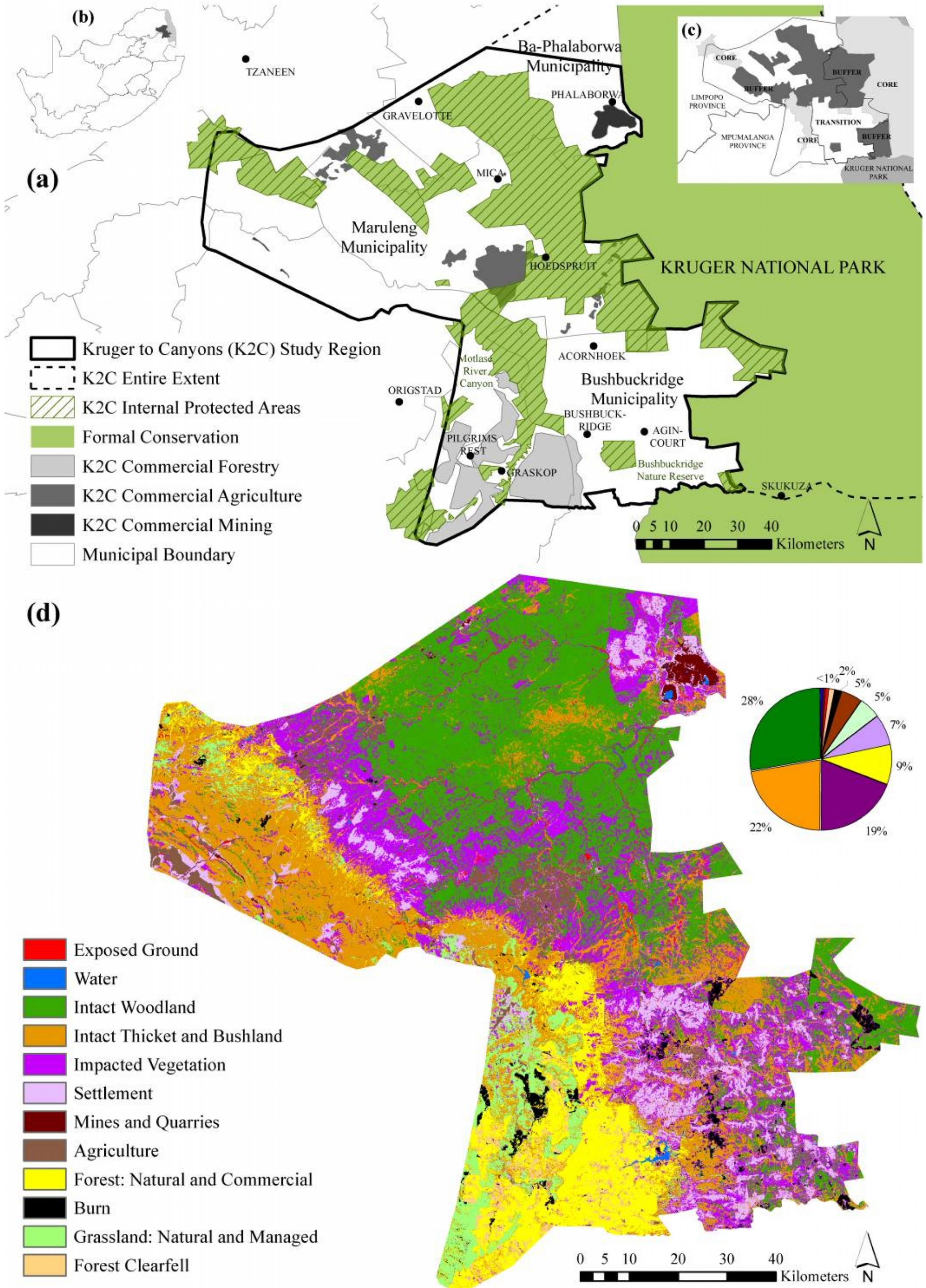


Figure 1 a) Regional location of the Kruger to Canyons Biosphere Reserve (K2C), including land-use, administrative boundaries and main towns. *Inset* (b) indicates location relative to national and provincial boundaries, *Inset* (c) spatial arrangement of core, buffer and transition zones. Fig. 1 (d) Updated 2012 land-cover classification map for K2C, inset indicates the percentage contribution of cover classes to K2C full extent.

The spatial zonation of all BRs is intended to minimise spatial conflict between environmental protection and environmental resource use at all levels, i.e. local household harvesting to commercial-scales of extraction (Coetzer et al., 2013a). While K2C's spatial zonation is not the concentric rings of stylised / model BRs, its spatial zonation, specifically core and buffer zones, is well-established in this landscape. Nationally protected PAs delineate the *core* zone, with the privately-owned PAs representing the *buffer* zone of K2C (Fig. 1c). This PA network accounts for more than half the available land area in K2C, and given its substantial spatial investment, land-use intensity in the unprotected remainder, the open-ended *transition* zone ('zone of co-operation') is significant (Coetzer et al., 2013b).

5.3.2 Data sources & analysis

5.3.2.1 Data acquisition and data processing

The 2012 land-cover map was created from winter Landsat^{ETM} *SLC-off* imagery acquired from The U.S. Geological Survey (USGS, www.usgs.gov), updating the classification time series from previous studies over this study region. Image acquisition dates were matched as closely as possible to reduce seasonal effects between *i*) tiles in 2012 image pair, as well as *ii*) the scenes from the original 1993 and 2006 time-series.

Fig. 1d presents the updated 2012 land-cover classification map, including the relative percentage contribution of the thirteen classes to landscape extent. Preliminary accuracy assessment (i.e. in the absence of extensive field surveys, but performed using complementary remotely-sensed data sources) suggests overall map accuracy of 84%.

Raw images were processed to Standard Terrain Correction (Level 1T: precision and terrain correction; www.usgs.gov for more information on USGS Landsat processing details, including ground-control points: *Global Land Survey 2000* and Digital Elevation Model utilised: *SRTM: Shuttle Radar Topography Mission* and *GTOPO30: Global 30 Arc-second Elevation* include African coverage).

Scenes were ortho-rectified in ERDAS IMAGINE 13.0 (Intergraph Corporation, 2012) using the original 1993 and 2006 scenes as reference. Third order polynomial and Nearest Neighbour resampling options were specified as geometric model and geo-correction preferences.

5.3.2.2 *SLC-off data considerations*

Landsat *SLF-off* imagery refers to a permanent failure of the Scan Line Corrector (SLC) on the Landsat 7^{ETM} sensor that occurred 31 May 2003. This failure has caused all subsequent Landsat^{ETM} imagery to display wedge-shaped data gaps that increase in size near the edges of the scene (Storey et. al., 2005), and vary latitudinally from scene-to-scene.

The SLC is an electro-mechanical component that controls the forward motion of the Landsat satellite (USGS, 2003). When operating correctly, the SLC allows for parallel scans by the ETM scanning mirror, providing regular and contiguous coverage of the scene-space below. However, since the terminal failure of the SLC, instead of aligning each individual scan-line parallel to the previous one, the sensor now scans in a 'zigzagging' motion, alternating scan-to-scan overlaps with scan-to scan data gaps / 'underlaps' (USGS, 2003; Storey et. al., 2005), which increase with distance from the scene nadir (Fig. 2 presents this graphically).

This has resulted in an approximate 22% data loss along the edge of the scene, but with near contiguous coverage in the centre of the image (Storey et. al., 2005, also see www.landsat.**usgs**.gov for more information).

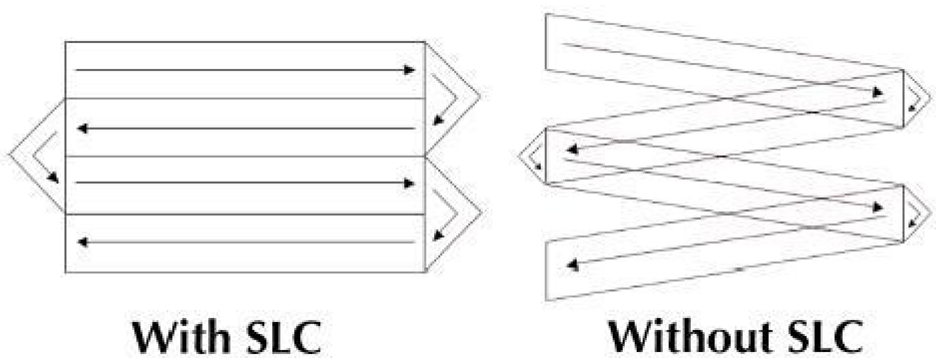


Figure 2. Stylised representation of Landsat^{ETM} scanning movement indicating SLC function: with a functional SLC (pre-May 2003) versus scanning motion following SLC failure, where the wedge-shaped data gaps are evident. ***Figure credit****: U.S. Geological Survey; http://landsat.usgs.gov/products_slcoffbackground.php*

There has been substantial international research into the most appropriate mechanism for filling these data gaps effectively, as the utility of SLC-off imagery remains considerable. This was

initially driven by the USGS, NASA, and the Landsat 7 Science Team in 2003, (USGS, 2003), but over a decade later, this research continues (e.g. Storey et. al., 2005; Zhang et al., 2007; Pringle et al., 2009; Chen et al., 2011; Zhu et al., 2012; Zeng et al., 2013).
Reasons for continued research interest include *inter alia*, *i*) with the exception of the data gaps, SLC-off data captured are comparable in radiometric and geometric quality as the data collected prior to the SLC failure (www.landsat.usgs.gov), *ii*) concerns relating to the age of *original alternative* sensor, Landsat 5, which was only recently decommissioned in June 2013 (launched in March 1984, with an original design life of 5 years,), *iii)* the lack of a continued global coverage of Landsat 5 data (Wulder et al., 2011) and *iv)* weather effects, i.e. clouds, necessitating an alternative source for certain dates / scenes even where global coverage of Landsat-5 prevails, and *v)* the *new alternative*, the replacement sensor Landsat 8, was launched in February 2013 (Landsat Data Continuity Mission (LDCM)) and only began normal operations in May 2013, when it was transferred from NASA to USGS (http://www.nasa.gov/).

The literature indicates three general categories for data reconstruction (Desai and Ganatra, 2012): *single source* (use of same image to predict gaps through pixel similarity / dissimilarity, and the averaging of missing pixels with those adjacent to them), *multi-source* (use of multi-date images of the same scene, with acquisition dates closely matched for image compositing (Wulder et al., 2011)) and *hybrid* (a combination of both approaches).

For the purpose of this study, a data fill technique that falls with the multi-source category has been utilised, using gap filling software distributed by NASA (‘*Frame and Fill’*). ‘Fill’ scenes are within 16 days of the primary / anchor image to negate seasonal effects and minimize any land-cover change resulting from short-term management interventions. Fill scenes were selected on the basis of their ‘fill potential’ and the latitudinal placement of data gaps relative to those of the primary scene.
Given the placement of the study area relative to the data gaps, and the natural overlap of the 2012 image pairs, the data fill procedure ensured that the residual data loss to the SLC-anomaly was negligible. Following manual reclassification of the remaining data errors in the final 2012 classification map, data gaps affected <0.01% of the classified area.

5.3.2.3 Classification procedure

The 2012 land-cover classification procedure mirrored that performed for the original 1993 and 2006 land-cover maps (Coetzer et al., 2010). The supervised classification approach was performed in IDRISI Selva (Eastman, 2012), assigning equal prior probabilities to the land-cover classes. Extensive manual reclassification and post-classification expert refinements of the classification output were performed in ArcMap 10.0 (ESRI, 2010), improving the quality of the classification product.

KNP has been excluded from all analyses, and only the land-cover of the matrix areas beyond the Park's fence-line is presented. Thus, the core zones referred to throughout the study refer only to the internal core zones, rather than the full extent that includes central KNP and the private PAs now fenced within its boundary.

Classification structure reflects broad land-cover types present in this landscape, with naming conventions that follow those of the National Land-cover (NLC) classification scheme (Thompson, 1996). Thirteen cover-classes were defined at the highest level of the classification hierarchy. Coetzer et al. (2010) describes the classes in detail.

This chapter focused on the spatial change dynamics of the three consolidated land-cover classes, as originally presented in Coetzer et al (2013b): combined Intact Vegetation, combined Impacted Vegetation and the Settlement classes. Hereafter these are referred to as the 'priority classes'. These priority classes are intended to portray the environmental protection versus environmental utilisation aspect of the theoretical BR model as has been applied in K2C, and account for >75% of the land-cover across K2C in all three time intervals.

5.3.2.4 Intensity Analysis

The changes in these priority classes have been characterised using the *Intensity Analysis*. The mathematical framework quantifies the changes in categorical data, i.e. classification maps, across multiple time intervals simultaneously, with analysis considering multiple levels of observation, i.e. from the landscape scale to the transitions *within* and *between* classes. The rationale behind this approach is that typical approaches to change detection do not indicate whether observed changes are derived from processes that are systematically more or less intensive in certain areas of the landscape compared to those that are uniform or random over time (Aldwaik and Pontius, 2012). As a result, such an approach provides insight into the underlying causes / processes driving the transformations between cover classes at different times (Huang et al., 2012).

Pontius et al. (2004) provides the seminal work that evolved into what is currently being applied as Intensity Analysis globally (Aldwaik and Pontius, 2012; Intensity Analysis website: https://sites.google.com/site/intensityanalysis/).

The original methodology considered a single time interval and analysed the size of the changes in land-cover categories relative to the overall landscape change, identifying the signals of systematic versus random transformation between the two time points (Aldwaik and Pontius, 2012). The current application considers at least three time points concurrently, and synthesises three levels of analysis (interval (general), category and transition (specific); Fig. 3) to detect the shift in landscape patterns through the time-series. At each level, the approach determines stationarity of patterns, i.e. whether observed changes are consistent over time, accounting for the difference in length of the study time intervals (Aldwaik and Pontius, 2012).

For the purpose of this study the focus is on the interval and category levels.

The *interval* level assesses total change in each time interval, determining the variations in total size and the intensity of rate of annual change across each time period. The observed rates are compared to a uniform rate, i.e. the rate that would exist if annual changes were distributed equally across the study's full temporal extent, with a relative measure of time intensity, 'fast' or 'slow', indicated (Fig. 3). The *category* level assesses changes in gross gains and losses across the landscape within each category, identifying the categories that are relatively active versus dormant in terms of spatial change over the specified time interval. The category intensity compares the observed intensity of change with uniform change, i.e. the change that would exist if category-level changes within each time period were distributed uniformly across the landscape. The *transition* level is the final level of analysis. Herein transitions between categories are analysed, providing insight into whether categories are actively 'avoided' or 'targeted' by other categories as they gain or lose in transformation. The observed intensity of each transition is compared with a uniform intensity that would occur if transitions were distributed equally amongst the land-cover categories available for transformation (for more detail, as well as the equations that form the basis of the Intensity Analysis, see Aldwaik and Pontius, 2012).

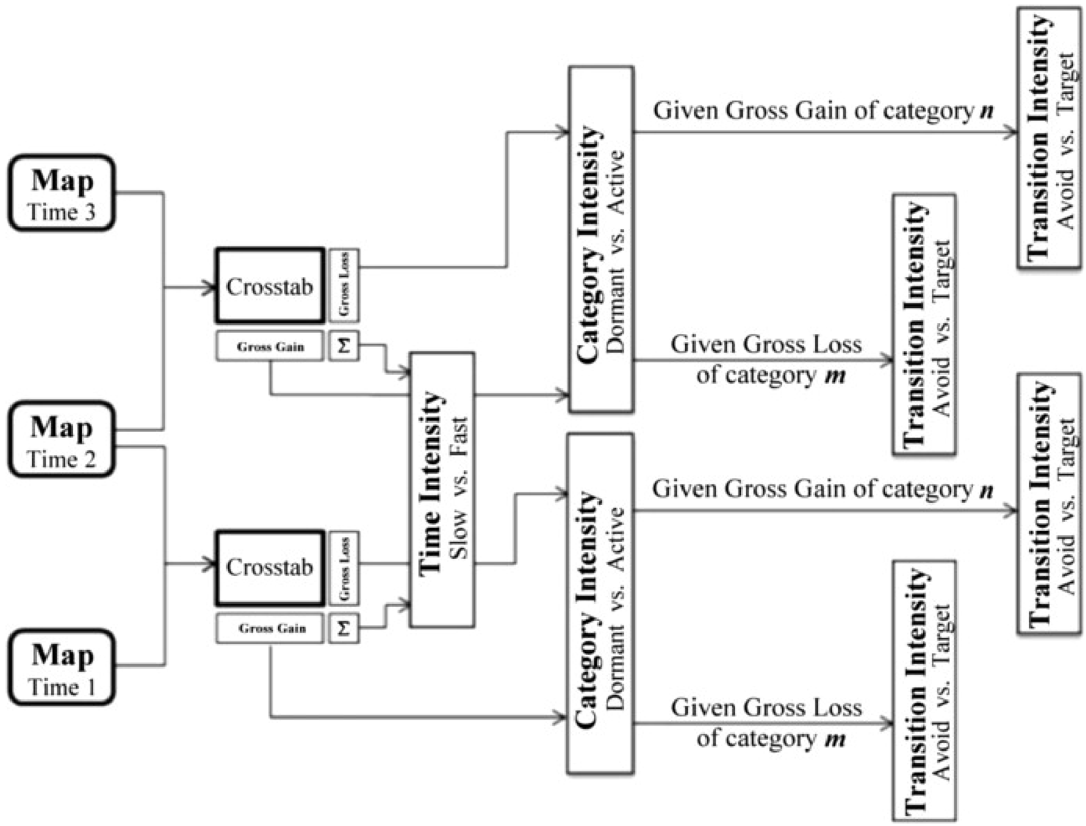


Figure 3. Intensity Analysis methodological process across analysis levels. ***Figure credit:*** *Aldwaik and Pontius (2012)*

Inputs for the analyses are cross-tabulation matrices between time points for consecutive time intervals, i.e. in the case of this study, 1993 and 2006, 2006 and 2012. Time intervals are not restricted to two time intervals as presented in this chapter, nor are the analyses restricted to a set number of land-cover classes - although three classes have been shown to increase ease of interpretation and/or demonstrate the concept of Intensity Analysis with clarity (Pontius et al., 2013).

In this study four cover classes are included, the three priority classes, as well as an additional combined class of ‘Other’, that accounts for transitions that are not restricted to exchanges between the priority classes. However, interpretation of these exchanges with 'Other' will be limited, given the aggregation of multiple classes into a single '*not-priority cover*' category.

There is some overlap between Intensity Analysis and the more typical Markov approaches (and spatial variations thereof, presented in the section below; see Aldwaik and Pontius, 2012; Pontius et al., 2012). However, the Intensity Analysis approach is not intended to replace Markovian analyses. Rather it complements the answers that a Markov transition probability matrix can provide, including an additional multi-point temporal assessment of change.

5.3.2.5 Simulation of future changes

To investigate future land-cover change in priority classes, a combined Cellular Automata – Markov chain model (CA-Markov) based on the 2006 – 2012 analysis interval has been utilised. CA-Markov approaches are regularly used in land-cover modelling, and common in the literature (reviewed in: Parker et al., 2003; Brown et al., 2004; Brown et al., 2007; Memarian et al., 2012), with proven applicability for modelling ecological aspects of land-cover change (Parker et al., 2003).

Typical Markov approaches have been criticised for ignoring spatial relationships between land-cover categories, and assuming the spatial independence of transitions over time (Brown et al., 2000). The inclusion of a cellular-automata (CA) component to the model addresses this limitation, allowing the transition probabilities between cells in the classification map to be a function of their neighbours (Pontius and Malanson, 2005), allowing for likely / realistic spatial transformations. Thus the hybrid CA-Markov model retains the transition probabilities of the Markov approach (control of the temporal change dynamics), but includes control of the spatial dynamics through the CA, i.e. spatial proximity (Houet and Hubert-Moy, 2006).

Changes for *2012 +6 ye*ars into the future, i.e. 2018 have been simulated, on the basis that this final time point is equidistant from the 2012 update, with a time interval (12 years) that complements the original 1993 – 2006 analysis period (13 years). Initially longer time trajectories, +10, +20 years were explored, but with increasing time since the input of the validated classification map, the CA-Markov decision-making became spatially more unrealistic for this landscape. This decision is based on the land-cover change 'behaviour' for intervals 1993

– 2006 (Coetzer et al., 2010; 2013b), and 2006 – 2012 (Results, this chapter), and expert-knowledge of this landscape.
This observation is supported by other studies (Falahatkar et al., 2011, Samat et al., 2011): over long time intervals the assumption that future trends will match past trends becomes increasingly flawed, impairing model predictions.

I acknowledge the limitations of CA-Markov / Markov approaches in general (reviewed in Memarian et al., 2012), as well as those linked to this data-set, i.e. predictive power of economic or population models have not been incorporated, nor have landscape suitability measures been applied, such as land-tenure and associated management strategies, or planned development to guide iterations. Rather, this simulation has assumed a purely spatial perspective and a homogenous rate of change across the landscape, ignoring administrative boundaries.

Model validation is infrequently reported in the literature (Pontius et al., 2001), and while this prediction can only be compared to actual land-cover distribution in 2018, the 1993 – 2006 simulation of possible 2012 land-cover has been utilised as a proxy measure for the accuracy of the 2018 output (see appendices).
The CA-Markov simulation output is not intended to be a 'hard' prediction surface for K2C, rather it has been used to suggest *a single potential future classification surface,* e.g. one scenario of the possible land-cover distributions that may occur, in the absence of any intervention if transformation trends continue in the short-term.

5.4 Results

5.4.1 Patterns of change in priority land-cover classes across K2C: 1993 – 2012

The priority classes accounted for 78 – 80% of the K2C landscape at all three time points, with the general trends presented in previous studies (Coetzer et al., 2010, 2013b) enduring with the addition of 2012 land-cover (Fig. 4).

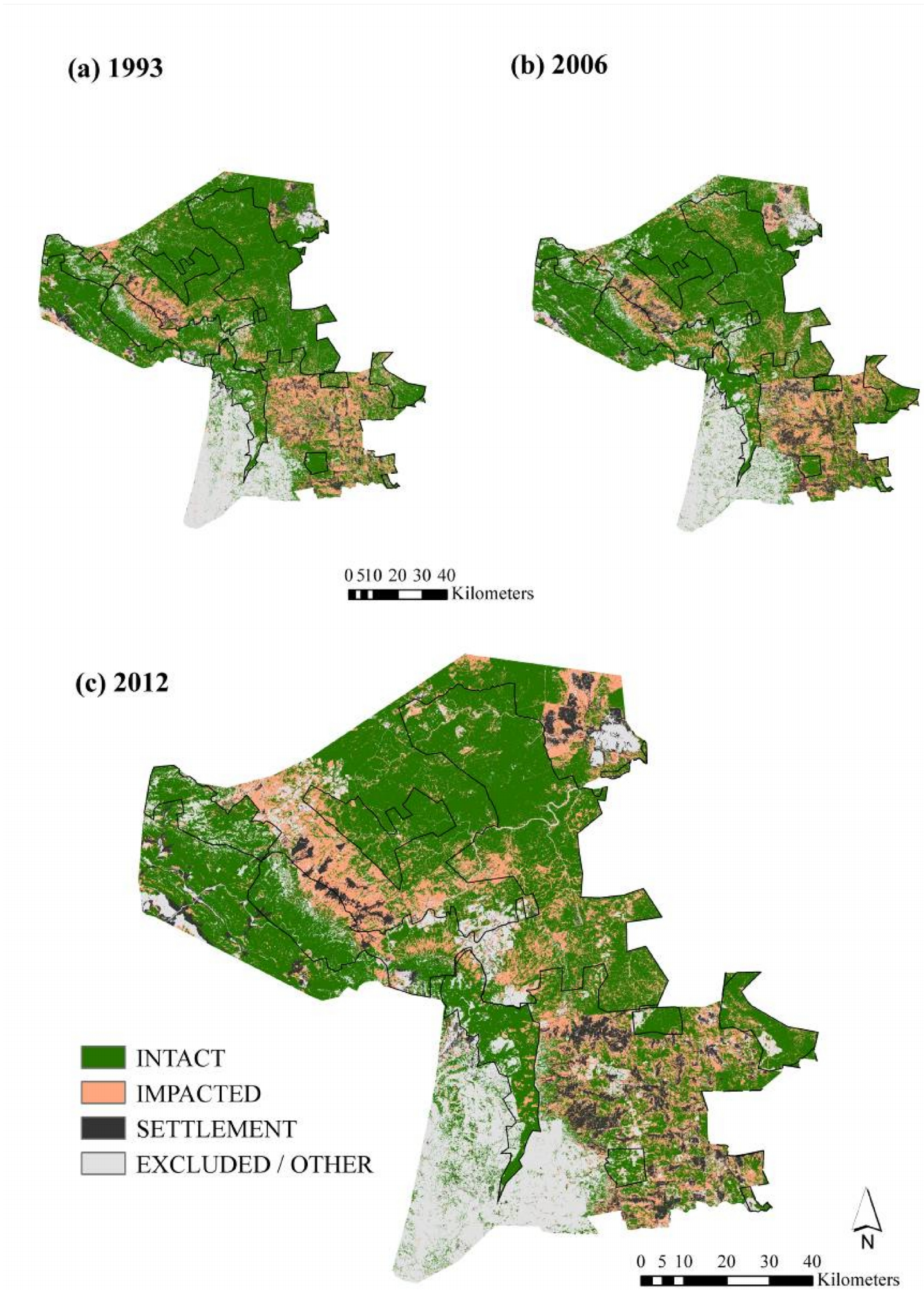


Figure 4. Spatial distribution of Intact Vegetation, Impacted Vegetation and Settlement classes (the 'priority classes') in the Kruger to Canyons Biosphere Reserve (K2C) relative to spatial zonation for a) 1993, b) 2006 and c) 2012.

Interpretation note: Years 1993 and 2006 are included for visual comparison, originally presented in Coetzer et al. (2013b)

Spatially, the majority of the change evident in the three priority classes predominates in the transition zone of the Biosphere Reserve, exterior to the spatial restrictions on use / large-scale land-cover change imposed by the ‘hard’ fences of the core and buffer zone’s PAs. In this regard, Impacted Vegetation has made noticeable advances in central K2C, encroaching towards the fences of buffer zone PAs here.

Intact Vegetation remains the dominant land-cover class in the BR in 2012, accounting for 52.17% of the landscape (Table 1). This reflects a continued decline in this cover type from 1993 to 2012: a total >7% decline relative to K2C’s total extent from 1993, and >3% decline from 2006. However, this change considered as a percentage relative to the class’s original 1993 and 2006 extent is more concerning: 6.3% loss from 2006-2012 (~>350 km^2), and 14% loss over the entire study interval (~>750 km^2).

Impacted Vegetation and Settlement classes have continued to expand across K2C. By 2012, Impacted Vegetation accounted for 19.1% of the subregion, while Settlement had expanded across 6.97% of K2C. This is an increase of 2.9% (~>330 km^2) and 2.8% (~>300 km^2) over K2C since 2006 respectively, and translates to expansion relative to their 2006 class extents of 10.2 and 16%.

From the transition matrix (Table 1) it is evident that the largest changes within both time intervals are exchanges between Intact Vegetation and Impacted Vegetation cover. Interestingly, exchanges from Intact Vegetation to the ‘Other’ classes in the landscape are also noticeable in both time intervals (>5% of K2C). Coetzer et al. (2013b) provides explanation for the exchanges from 1993 – 2006: with transformation to Cultivated Land a substantial contributor. For the 2006 – 2012 period, upon detailed inspection of the 2012 land-cover classification map (Fig. 1d), additional losses of Intact to burned areas have become more prominent in the landscape. However, whether these translate into ‘real’ losses over time (i.e. short-term conservation management interventions at local scales that are typical of this landscape, versus a mechanism of clearing for agricultural purposes), will need reassessment with subsequent land-cover analyses.

Table 1. Land-cover transition matrix for priority cover classes (measured as percentage of K2C): a) 1993 – 2006 and b) 2006 – 2012

Interpretation note: Percentage values on the diagonal (highlighted in dark grey) indicate the percentage of the cover class that has persisted over the time interval. Values presented in end columns / rows (highlighted in light grey) indicate the total coverage of cover class in each time interval. Percentage values in remaining cells (no highlight) indicate the exchanges between cover classes, with earlier analysis year presented on the vertical.

a)

	Cover class	**2006**				*Total 1993*
		Intact	Impacted	Settlement	Other	
1993	Intact	45.27	7.69	0.91	5.61	59.49
	Impacted	4.84	7.64	1.98	1.70	14.46
	Settlement	0.70	0.61	2.33	0.56	4.20
	Other	4.64	1.27	0.64	13.72	20.26
Total 2006		55.45	17.21	5.85	21.59	*100.00*

b)

	Cover class	**2012**				*Total 2006*
		Intact	Impacted	Settlement	Other	
2006	Intact	40.73	8.57	1.05	5.12	55.45
	Impacted	7.04	7.17	1.17	1.76	15.38
	Settlement	0.34	1.26	3.68	0.57	4.20
	Other	4.06	2.10	1.07	14.30	20.26
Total 2012		52.17	19.10	6.97	21.76	*100.00*

From a theoretical conservation viewpoint, it is encouraging that Impacted Vegetation may potentially return to a state structurally similar to what has been classified as Intact habitat (in so far as it is distinguishable at the Landsat resolution, see Coetzer et al. 2013b, specifically the structurally densely wooded Intact Bushland and Thicket cover class). However, I question the validity of a homogenous and an actual ecological or functional transformation from Impacted to Intact at the scale of this landscape, across all zones, without additional research.
Anecdotal observations from areas within the transition zone, suggests the densification of woody vegetation, increases in biomass and bush encroachment of certain species at specific local sites. Similarly, there is evidence from the literature that demonstrates that coppice growth as a result of intensive resource harvesting changes tree structure to a denser shrub-like growth

form, which at larger scales might be considered as bush-encroachment (Fisher et al., 2012). However, whether this is a wide-scale occurrence *or* can be effectively ‘up-scaled’ across different resolutions, beyond specific socio-economic contexts and village sites, requires further critical review before generalisations can be considered correct.

5.4.2 CA-Markov: Simulations of future changes

The CA-Markov simulation extends the spatial patterns of change in the priority classes to 2018. Figure 5 indicates the potential spatial change, determined on the basis of the Markovian probability matrix (the likelihood of persistence over the analysis period) and the 2012 spatial distribution of cover across K2C. Most noticeable is the spatial expansion and consolidation of Impacted Vegetation in the Poverty Nodes and around Palabora mine. By 2018, the simulation predicts that Impacted Vegetation will have increased coverage over K2C by almost 4% (Fig. 6), which combined spatially is an area of >400 km^2 and larger than the expansion during the previous 6 year period. Similarly, the model predicts the expansion of the Settlement class within the Poverty nodes, with settlement patch shapes becoming more compact units. However the 1% increase by 2018, is lower than what occurred from 2006 – 2012, equivalent to an area >100 km^2 across K2C.

Intact Vegetation continues to decline, with the simulated losses encroaching into central K2C, and reducing the number of remnant patches within the Impacted Vegetation, i.e. with the communal rangeland expanse. The model predicts losses of 5.1% of Intact Vegetation across the BR, an area equivalent to >500km^2, greater than what was transformed in both the 1993 – 2006 and 2006 – 2012 analysis periods.

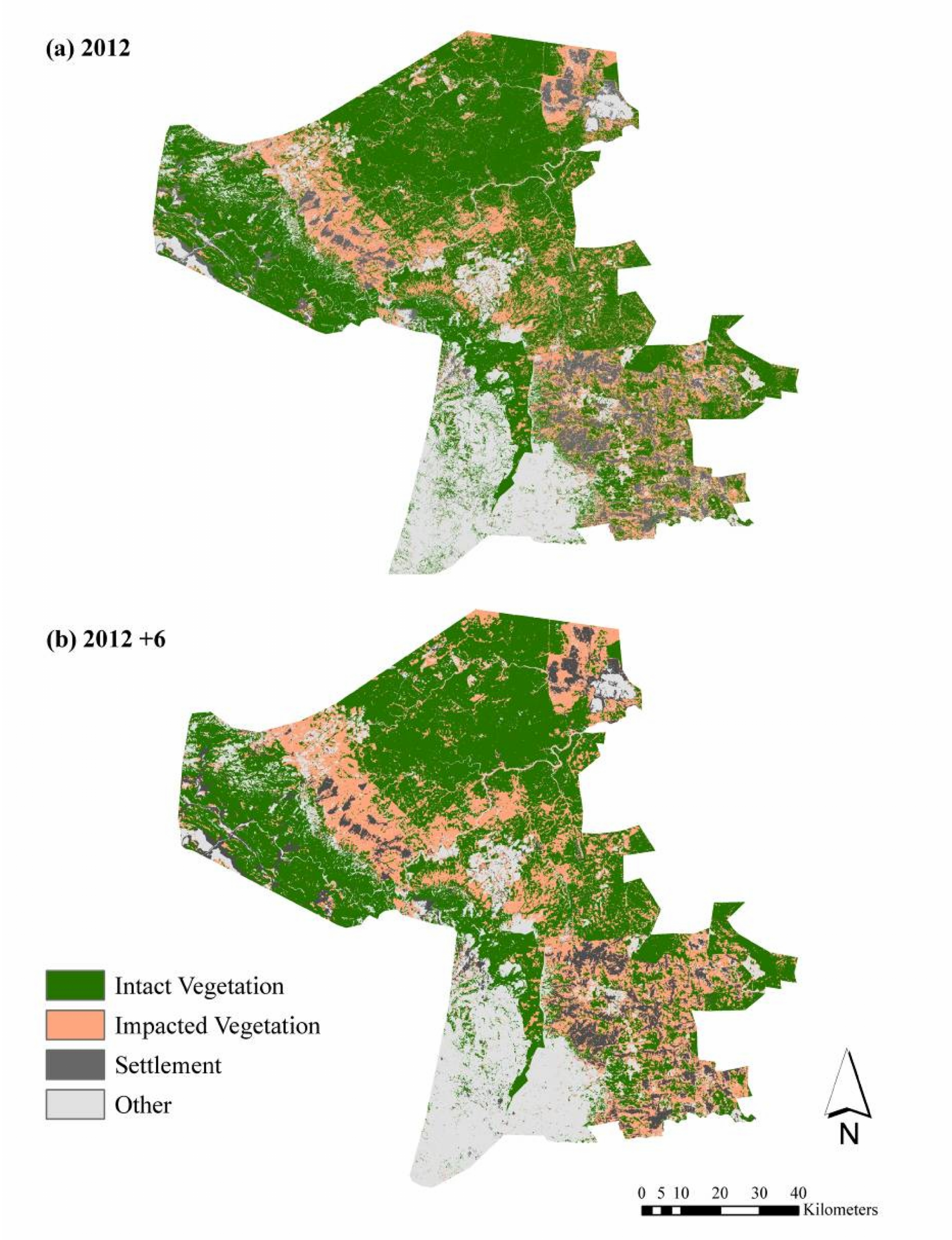


Figure 5. CA-Markov simulation of future land-cover changes for the priority cover classes for 2012+6. *Inset a)* includes the original 2012 distribution of priority classes for ease of comparison.

Note: Markov transition probability matrix, on which the simulation is partially based, is presented in Appendix 1, together with associated spatial representation of conditional probabilities for the *2012+6* simulations period; Appendix 2.

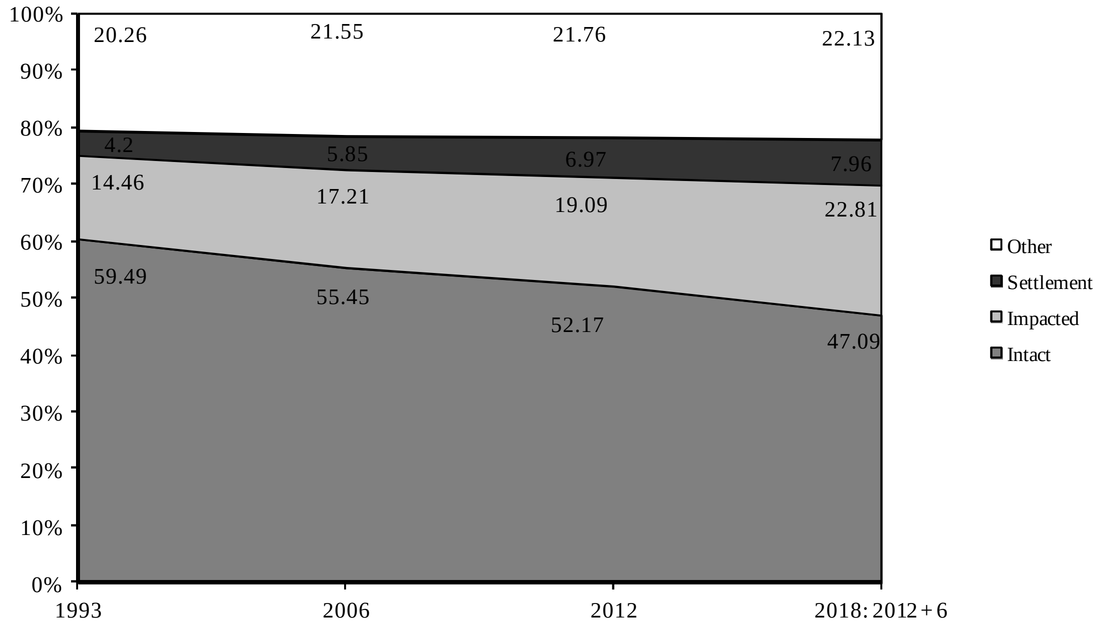


Figure 6. Change in percentage contribution of priority classes across the K2C landscape with the addition of the Ca-Markov simulation for the future 2012+6 time period. Included for ease of visual comparison are the spatial contributions of priority classes for original 1993, 2006 and 2012 classification maps (refer to Table 1).

5.4.3 Intensity Analysis: assessment of the annual rate and size of change across K2C

Table 1 does not however explicitly indicate whether the observed changes are accelerating or slowing over the entire analysis period, instead indicating the size and direction of net change. Nor does it reveal if the size of the observed change is a function of the spatial extent of the cover class, rather than an intensification of transformation (Pontius et al., 2004): larger land-cover categories may automatically have more expansive footprints of change, which may not necessarily equate to urgency in management action. This type of information is especially useful for the scheduling of practical or policy interventions, indicating intensive versus systematic exchanges between land-cover types (Alo and Pontius, 2008, Huang et al., 2012), and identifying critical transitions in land-cover categories over time. These ‘critical transitions’, i.e. faster or slower changes than anticipated for the landscape, could potentially act as an ‘early warning’ signal for transformation, and will have important management considerations for this, and other, landscapes.

Figure 7 presents the difference in intensity of landscape change within K2C over the two time intervals. Overall landscape change has accelerated from 2006 – 2012, with the annual change area increasing to 5.68% of the subregion, faster than what would be expected if changes were constant from 1993 to 2012.

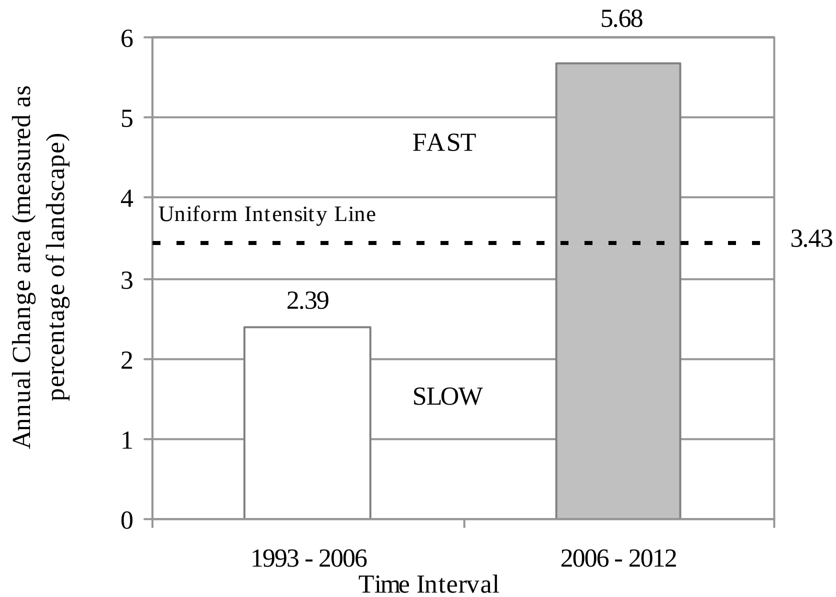
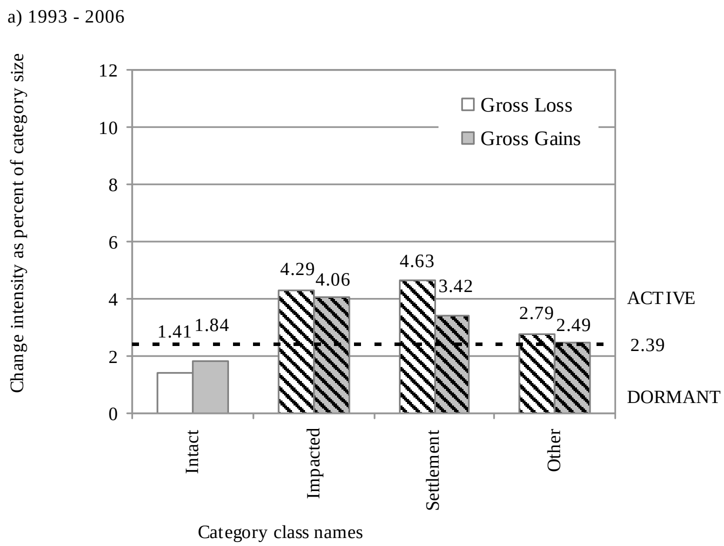


Figure 7. Intensity of change across the Kruger to Canyons Biosphere Reserve (K2C), presented by time intervals

Interpretation note (after Huang et al., 2012): The horizontal dashed line indicates 'uniform rate of change' that would occur if observed changes were distributed equally over time. If the time interval's annual change area is higher than this horizontal, (highlighted here in grey) then the change is relatively fast over that period, while lower levels indicate slower changes.

Category change intensities across time intervals show this increase as a function of the priority land-cover classes across K2C (Fig. 8). While these gross changes do not expose directional changes clearly, they do demonstrate a land-cover class's *annual intensity of activity* across the landscape across time intervals



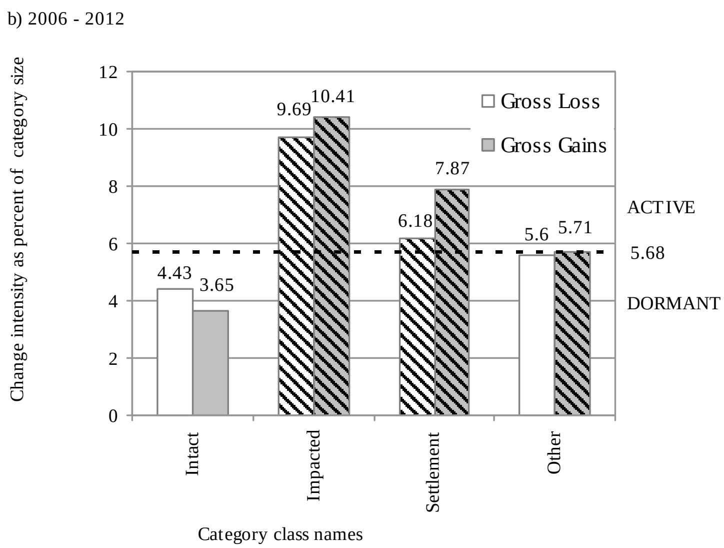


Figure 8. Category intensity changes for gross losses and gains across the Kruger to Canyons Biosphere Reserve (K2C) for a) 1993 - 2006 and b) 2006 – 2012

Interpretation note (after Aldwaik and Pontius, 2012; Huang et al., 2012): Category intensity indicates the relative size of gross gains and losses across categories, as a percentage of the category. The horizontal dashed line indicates the uniform intensity (%) of annual change for the entire landscape. This value differs between analysis intervals presented here as it is dependent on the area of change during the interval, as well as the duration of the interval. If the category change (gross gain / loss) extends beyond this horizontal then the change experienced by that cover class category is relatively active for that category compared to the landscape (highlighted by hatching / black diagonal 'fill' in both gains and losses). If the category change remains below the horizontal, then the change is relatively dormant for that category. If the intensity of changes were distributed uniformly across K2C, the bars would match the uniform intensity percentage value.

With the exception of Intact Vegetation in both periods, the land-cover classes show gross changes that are ‘active’ relative to the size of the class over time, accelerating accordingly in the 2006 – 2012 period, with gross gain intensities becoming more prominent in the landscape over time (Fig. 8b).

This indicates that for both Impacted Vegetation and Settlement classes, given the sizes of these classes in the landscape, observed changes (the expansions, contractions and spatial ‘swaps’) are more intensive than if the overall changes were distributed uniformly across K2C from 1993 – 2012.

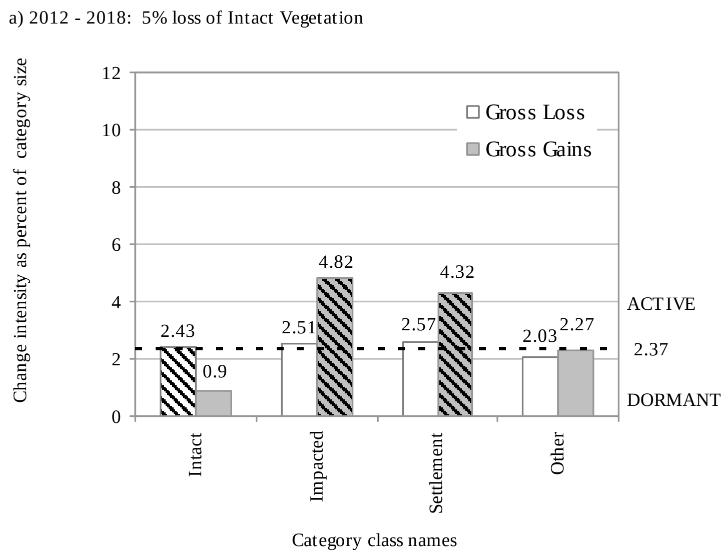
In contrast, Intact Vegetation annual change activity remains ‘dormant’ from 1993 – 2012. The observed changes relative to its spatial extent were lower than expected based on *i)* its proportional contribution to total K2C land-cover and thus, *ii)* the theoretical availability of this cover type for conversion. Regardless, the intensity of activity in Intact Vegetation is increasing over time across this landscape, with larger increases in loss intensity relative to gain intensity.

Figure 9a extrapolates this trend to 2018, based on the CA-Markov simulation of future change for this BR (Fig. 9a, 5%).

At the predicted 5% loss over a 6 year period, the Intensity Analysis predicts that the Intact Vegetation loss intensity makes the switch from dormant to active in this landscape. This suggests that a further 5% loss of Intact Vegetation may represent a critical threshold of change for K2C.

Manipulating the both percentage loss and the time in which this occurs within the Intensity Analysis procedure independently indicates the stationarity of loss activity. A further hypothesized 2% loss of Intact Vegetation beyond this over this same time period (Fig. 9b; 7%), and the intensity of loss increases accordingly, becoming more established on the ‘active’ side of the activity threshold.

However, should the loss of Intact Vegetation be slower than the simulation predicts, i.e. 5% over a 12 year period rather than 6 years (Fig. 10), Intact Vegetation change remains ‘active’ in this landscape. This reinforces the hypothesis that provided other factors remain unchanged, a further 5% loss of Intact Vegetation in this landscape is sufficiently large in size to affect change dynamics over short and longer temporal extents.



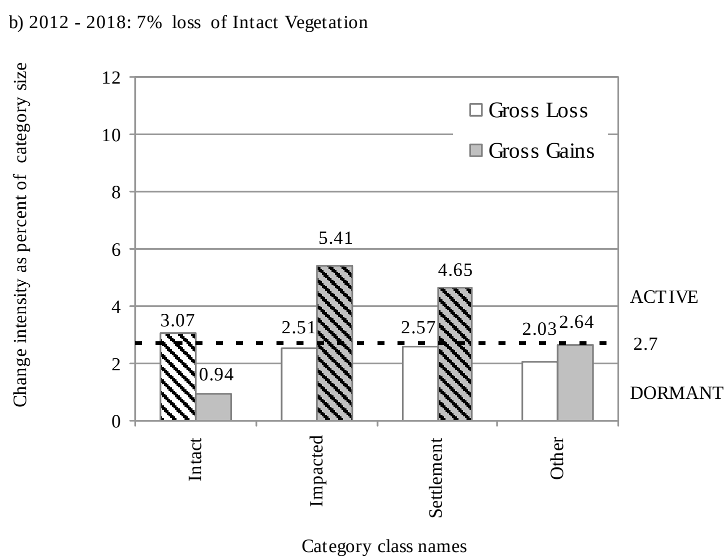


Figure 9. Hypothetical category intensity changes for gross losses and gains across the Kruger to Canyons Biosphere Reserve (K2C) for a) 2012 – 2018 prediction based on 5% future loss of Intact Vegetation, b) 2012 – 2018 prediction based on 7% future loss of Intact Vegetation.

Calculation note: For the 7 % loss, the additional 2% loss was assigned in the proportions as in the original transition matrix between the observed 2012 classification map and 2018 simulation.

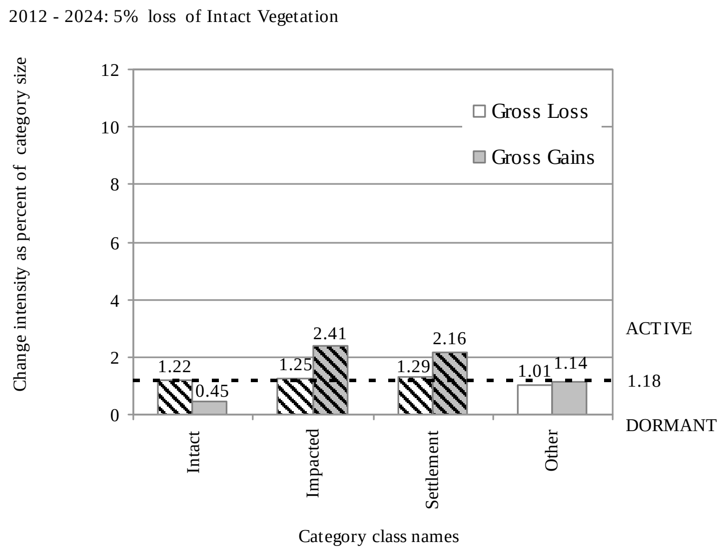


Figure 10. Hypothetical category intensity changes for gross losses and gains across the Kruger to Canyons Biosphere Reserve (K2C) for 2012+12 year scenario: 2012 – 2024 prediction based on 5% future loss of Intact Vegetation over this time.

5.4.4 Biosphere Reserve Zonation & Spatial changes: Gains, Losses and Persistence

The intensity analysis provides some insight into both the temporal change and spatial component of change across this landscape, which can be explicitly linked to the BR's zonation. Figures 10a,c,d demonstrate the spatial distribution of these gross changes for the 2006 – 2012 time interval: the gains, losses and persistence (GLP) over time for the priority classes. They are presented relative to the BR's spatial zonation, and include insight into the process of relocation across K2C, most noticeably the exchanges between cover classes.

Figures 10b,d,e indicate the percentage contribution of the GLP's relative to zone size, indicating the total class 'activity' within each zone over each time interval, and thus, the importance of each zone in the context of the gross changes that have been observed.

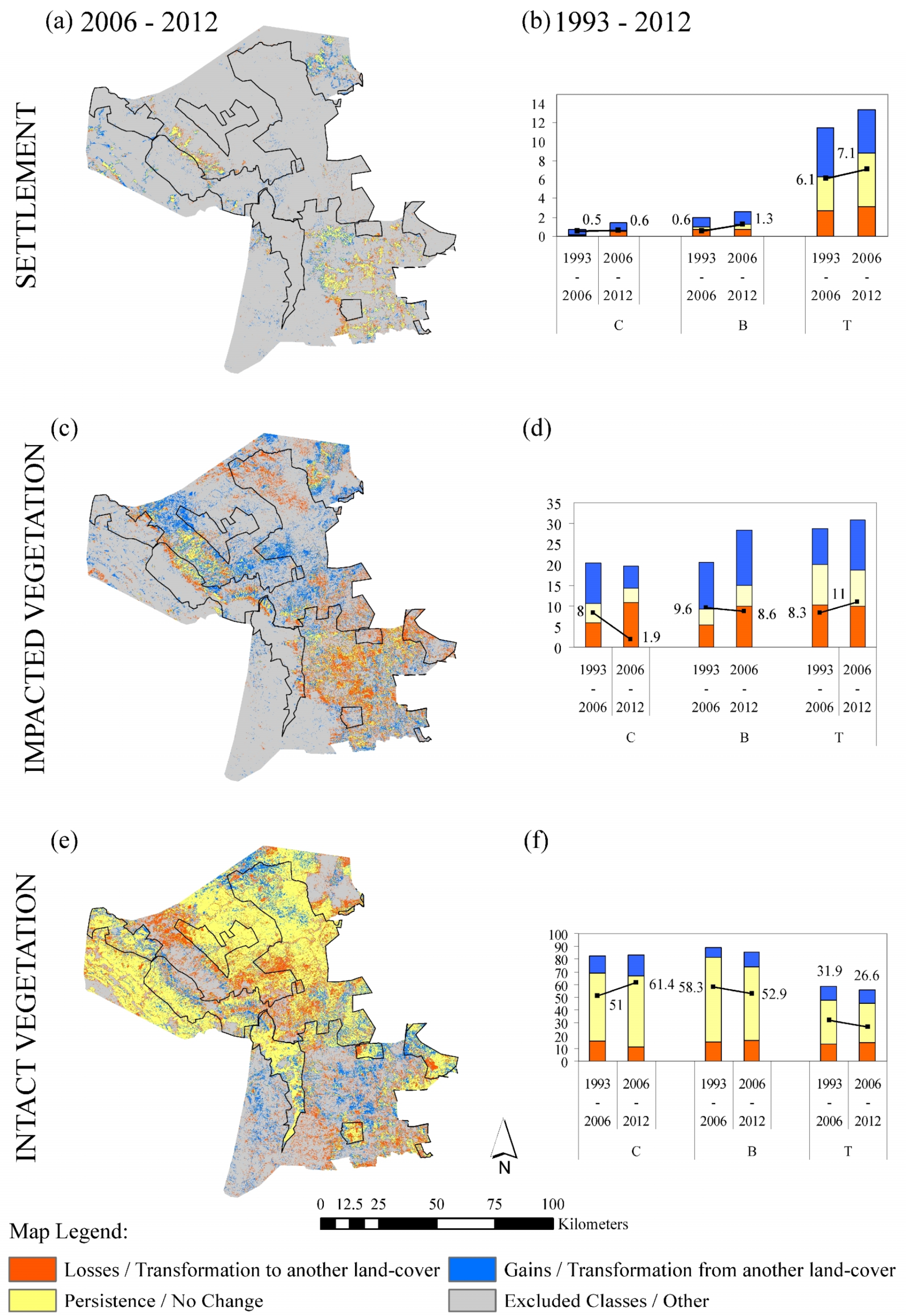


Figure 11. Spatial patterns of Gains (G), Losses and Persistence (P) in a) Settlement, c) Impacted Vegetation and e) Intact Vegetation for 2006 – 2012, with core and buffer zone boundaries indicated. *Insets b, d, f)* indicate these changes as a function of zone extent for 1993 – 2006 and 2006 – 2012.

Interpretation note: Solid lines within bars indicate the changes in class stocks from the end of each time interval for each zone, with percentage values of stocks for each period indicated, where C: Core Zone, B: Buffer Zone and T: Transition Zone. For visual interpretation the maximum percentage values on the y-axis differ between classes, with Settlement maximum: 16%, Impacted- and Intact- Vegetation 35% and 100% respectively.

At the level of BR zonation, the trends of change observed at the landscape level are not stationary across zones for all classes over both time intervals. The change in the standing stocks of Impacted Vegetation and Intact Vegetation at the end of each time interval (2006; 2012) for core and/or buffer zones indicate decreasing and increasing trends respectively (Fig. 11d,f), the reverse of what occurred across the entire K2C, (i.e. from Table 1).
This suggests that formal conservation efforts within certain core and buffer zone PAs have had some success in checking transformation within their borders, e.g. Molatse (formerly Blyde) River Canyon and the escarpment PAs (Fig. 1). I must caution that this change is not uniform across *all* PAs within K2C, as for example, Bushbuckridge Nature Reserve (buffer zone PA) continues to experience transformation within its borders (Fig. 11e), a scenario potentially worsened in the future with improved road access to areas of the reserve, with implications for resource harvesting by nearby communities.

Settlement has standing stocks that have increased across buffer and transition zones, (the core zone's contributions are negligible, and remain largely constant across time; Fig. 11b), with gains in settlement encroaching into areas of Impacted and Intact cover (Fig. 11a). Increasing standing stocks here are consistent with the landscape for the Settlement class (Table 1); landscape scale trends are largely governed by the spatial predominance of the Settlement class within the transition zone, for which the zone was originally delineated.

5.5 Discussion

The spatial trends of change across K2C have continued with the addition of the 2012 classification map to the 1993 and 2006 time-series (Fig. 4), with further increases in Settlement and Impacted Vegetation cover, and declines in Intact Vegetation across the BR (Table 1). While these general trends of change are consistent across both analysis intervals, the overall change has accelerated over time (Fig. 7), becoming noticeably more intensive around the population centres of the transition zone in the 2006 – 2012 analysis period, i.e. within Bushbuckridge, Maruleng and Ba-Phalaborwa Municipalities (Figs. 8 and 11).

The increases in population sizes for these municipalities since the previous analysis period, may explain some of this acceleration (Fig. 12), with the assumption that an increased population, contributes to a larger settlement expanse with associated greater harvesting areas. Previous work in this area has indicated the importance of the spatial relationship between a more extensive settlement expanse and the footprint of local environmental resource use (Coetzer et al., 2013b), a landscape observation supported by recent research conducted at village scales (Fisher et al., 2012; Matsika et al., 2013; Wessels et al., 2013), with more intensive utilisation compromising the ecological values of these harvesting areas.

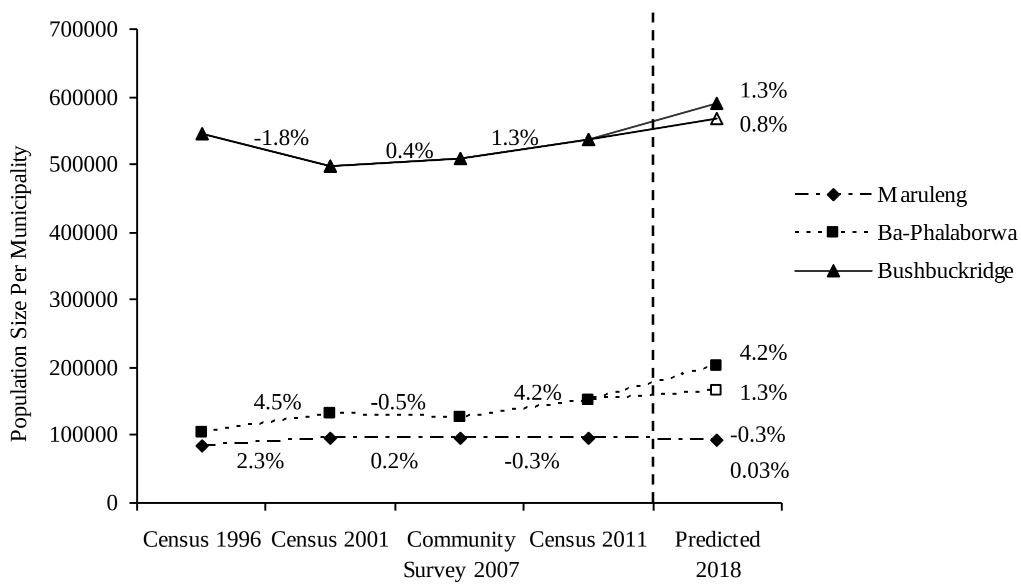


Figure 12. Human population sizes of Bushbuckridge, Maruleng and Ba-Phalabora Municipalities

Interpretation note: Percentage value positioned alongside the trend-line indicates population percentage growth rate (PGR) between observation periods. Dashed vertical line represents the prediction of population size in 2018 (2012+*6*), with extrapolations based on i) PGR between 2001 and 2011 national census data (hollow place-markers) and ii) PGR between 2007 Municipal community surveys and the 2011 census data (filled place-markers). Both PGR values are included right of the dashed vertical, alongside place-markers, with the census data-based PGR (i), the lower value in each pair. From the graph, the difference in predictions for Maruleng Municipality is slight, 2090 people, and thus not clearly evident from this graph, but total size predictions based on i) > ii).

However, the literature cautions against assuming that poverty and population growth are the primary drivers of land-cover change in rural landscapes (Lambin et al., 2001), as the ‘rich’, i.e.

measured at the scale of commercial forestry, mining, and agricultural operations have certainly had substantial impact on natural ecosystems throughout the developing world (de Sherbinin et al., 2008). While K2C is no exception, as these formal industrial economic sectors remain prominent in this landscape, accounting for ~>15% of the BR in 2012 (Fig. 1), they contribute to <2% of the total landscape change between 2006 and 2012. Rather the feedbacks between recent increases in population growth, (and likely future increases based on current trends, Fig. 12), decreases in household sizes (Bushbuckridge Local Municipality, 2013), coupled with the chronic scarcity of economic opportunities, and the legacy of historical policy and biased land tenure regimes (Carruthers, 1995; Pollard et al., 2003) has had significant consequences for land-cover change in this BR region (e.g. Banks et al., 1996; Twine et al., 2003; Madubansi and Shackleton, 2007; Matsika et al., 2013).

The neighbourhood of surrounding vegetation cover impacts the future potential of land-cover in a given area, with neighbouring cover not only important for local effects (e.g. edge effects), but also for influencing the broader spatial context of the landscape (Crews and Young, 2013), with initial transformation fuelling further transformation trajectories (Brady et al., 2009). Thus, the increased expansion of Settlement and Impacted Vegetation beyond previously observed localities are predicted to have substantial repercussions for the Intact Vegetation expanse in the near future, i.e. *2012+6* (Fig. 6), with simulated losses, in the absence of intervention, escalating spatially, and directly impacting the linkages between buffer and core zone PAs across K2C (Fig. 5).

The 1993 – 2006 model validation of future land-cover performed 'moderately' well for the distribution of Intact Vegetation, Settlement and Other classes across K2C in 2012, with Cohen's kappa statistic (*K*) values $\geq$ 0.60 (see Landis and Koch, 1997 for interpretation). However, the model has underperformed for the spatial prediction of Impacted Vegetation cover across the BR, with K<0.40, translating to only 'Fair' agreement with the 2012 classification map.
I argue that this can be explained twofold: *i)* the private PAs within the BR are known to engage in bush-clearing and/or mowing to combat bush encroachment (Fisher et al., 2014), which, when remotely-sensed, these cleared areas may mimic the spectral signatures of the more typical Impacted Vegetation of the communal rangeland areas (Coetzer et al., 2010). Artificial water

points may also facilitate similar changes in vegetation composition and vegetation clearing through enhanced grazing (Parker and Witkowski, 1999; Thrash, 2000; Smet and Ward, 2005; Smit et al., 2007). Such management strategies are unlikely to be well predicted by the simplistic simulation of land-cover distributions presented here.

Similarly, and perhaps more importantly for areas of the transition zone not under formal conservation management is that: *ii*) not all Impacted Vegetation cover is contiguous with large areas of pre-existing Impacted Vegetation, and as such, the CA-Markov model is unlikely to favour its spatial expansion. Our previous research has indicated that in addition to cohesive areas of conversion, transformation to Impacted Vegetation only required a small patch, a 'kernel', of existing Impacted Vegetation cover to initiate conversion (Coetzer et al., 2013b).The importance of these isolated, small patches is discounted by the model simulation.

Thus, I posit on this basis, that the model prediction of future cover, i.e. the 2018 simulation, will under-estimate the spatial distribution of Impacted Vegetation in the transition zone, with concomitant issues for the already declining stocks of remnant Intact Vegetation of these areas (Fig. 11f).

There is a growing community awareness of the loss of ecological function and degradation of communal rangeland areas due to increased harvesting demands (e.g. Kirkland et al., 2007) with villages annexing neighbouring areas to meet livelihood demands (Matsika et al., 2012), and the spatial consolidation of previously distinct rangeland areas (Coetzer et al., 2013b; Fig. 1d). Current concerns around resource scarcity have led to the formal fencing of individual rangelands to limit access and establish 'ownership' over resource use. These fences were not observed until recently (field observations), and is likely to have considerable repercussions for areas with lower levels of regulation. In this context, Bushbuckridge Nature Reserve is considered as an example.

Bushbuckridge Nature Reserve (BNR) is well positioned in central Bushbuckridge Municipality, midway between Kruger- and Molatse (Blyde) River Canyon- Nation Parks, providing the spatial linkage for a connected PA network in the southern part of the BR (Fig. 1a). It is a buffer zone PA, approximately 6800 ha in extent and under administration by the provincial

Mpumlalanga Tourism and Park Agency (MTPA), with a permanent environmental staff of twelve (one Reserve manager, ten field rangers, and a shared ecologist).

Over the analysis period, 1993 – 2012, results indicated increased transformation of cover at the Reserve's boundaries, with Impacted Vegetation encroaching into the core of the PA (Coetzer et al 2010; 2013b; Figs. 1d and 4), most likely due to a chronic lack of appropriate fencing (Bushbuckridge Local Municipality (BLM), 2013). This uncontrolled access has resulted in a reputation of a PA that is "poorly managed and considered dangerous" (BLM, 2010), and in urgent need of ecosystem rehabilitation (BLM, 2013).
Together with Andover Nature Reserve (buffer zone PA located on the northern boundary of the Bushbuckridge Municipality), BNR has been targeted for economic revitalisation, with infrastructural upgrades a priority (BLM, 2013). However, for BNR, any improvement made to roads in the absence of more rigorous entry regulations, effectively enhances access to previously unavailable areas of the Reserve, and will fuel more rapid transformation. Anecdotal evidence has already indicated increased resource harvesting in the Reserve.

With regard to the Intensity Analysis, should the trends of landscape change continue into the future as the simulation predicts, the point at which Intact Vegetation losses become 'active' in this landscape (Fig. 9), signals a theoretical shift in the landscape dynamics, and identifies an opportunity for proactive and precursory management intervention. For K2C, a 5% loss may represent a potential 'critical threshold of change' for scheduling action, which then can be directed at the appropriate BR zone (Figs 9a and 10).

On the basis of the results presented here, the remnant Intact Vegetation of the transition zone, and its role in facilitating connectivity across the BR, will most likely become a scheduling priority. While a substantial portion of K2C's Intact Vegetation remains sheltered from potential transformation (yet with disparities in protection effectiveness linked to regional administration (e.g. Limpopo versus Mpumalanga provinces) and land ownership (private versus provincial management); Fig. 11e)), half of K2C's remaining Intact Vegetation is located beyond existing PAs (51% in 2012), and thus available for conversion.

K2C stakeholders have already initiated measures to increase protection of threatened areas beyond the current core and buffer zones (http://www.kruger2canyons.org/). One such measure includes a proposal to expand the boundaries of a private PA on the eastern interface of Bushbuckridge Municipality and Kruger National Park; a phased expansion of 4500 ha into the Bushbuckridge transition zone, constituting a formal co-management agreement between the Sabi Sands Wildtuin (SSW) and the Amashangane tribal authorities.
The 2012 classification map indicates this area remains largely Intact Vegetation (Fig. 1d) and thus, a suitable option for expansion. However inclusion into formal conservation removes this land from communal utilisation, and as such, associated livelihood scenarios and community beneficiation frameworks are a necessity.

5.6 Conclusions

Time-series data has revealed new insight into the process of land-cover change across K2C, information that would have been overlooked had a two-point observation of change been retained. The increased rate of change in recent years poses challenges for landscape management, with future predictions of escalating transformation likely to undermine BR sustainability. Through the Intensity Analysis approach I have hypothesized that a further 5% loss of Intact Vegetation represents a critical threshold of change for this landscape, beyond which landscape change may accelerate more rapidly, with likely consequence for overall conservation values and system function across the BR.

I appreciate that the simulation of future losses of Intact Vegetation and the results of the Intensity Analysis do not completely identify with the reality of the existing land-use mosaic in the BR, i.e. the spatial distribution of PAs. Yet, such landscape analyses remain valuable for BR decision-making, as the ongoing integrity of the PAs will depend, at least in part, on the integrity of the land-cover that surrounds them (Jones et al., 2009; Svancara et al., 2009; Townsend et al., 2009).

For the period, 2006 – 2012, the transition zone suffered a loss of 2.5% of Intact Vegetation cover (25.6%$_{2012}$ vs. 28.12%$_{2006}$), and 4.17% loss since 1993 (Coetzer et al., 2013b: Table 5). As such, based on past trends, a 5% loss of Intact Vegetation in the next 6 years exclusively from the transition zone, may exaggerate the potential for transformation here. However, over the next 12 years, excluding the potential for change acceleration and extrapolating past losses, such a loss is not improbable. Within this extended temporal extent, this 5% loss still signals a theoretical threshold for change across K2C (Fig. 10).

The Intensity Analysis approach to understanding landscape dynamics is only one of many appropriate options utilised (e.g. Lambin et al., 2003; Brown et al., 2004; Verburg et al., 2004), each with limitations and assumptions inherent in their application. In this study the Intensity Analysis approach has been utilised to complement a more typical spatial analysis of land-cover change in K2C, with the switches in change activity used as an indication for scheduling management intervention. However, whether this 'activity switch' as has been interpreted here, is translatable beyond the theoretical, and represents a real, and significant, threshold change in terms of ecosystem function / resilience, requires further investigation.

5.7 Acknowledgments

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5.9 Appendices

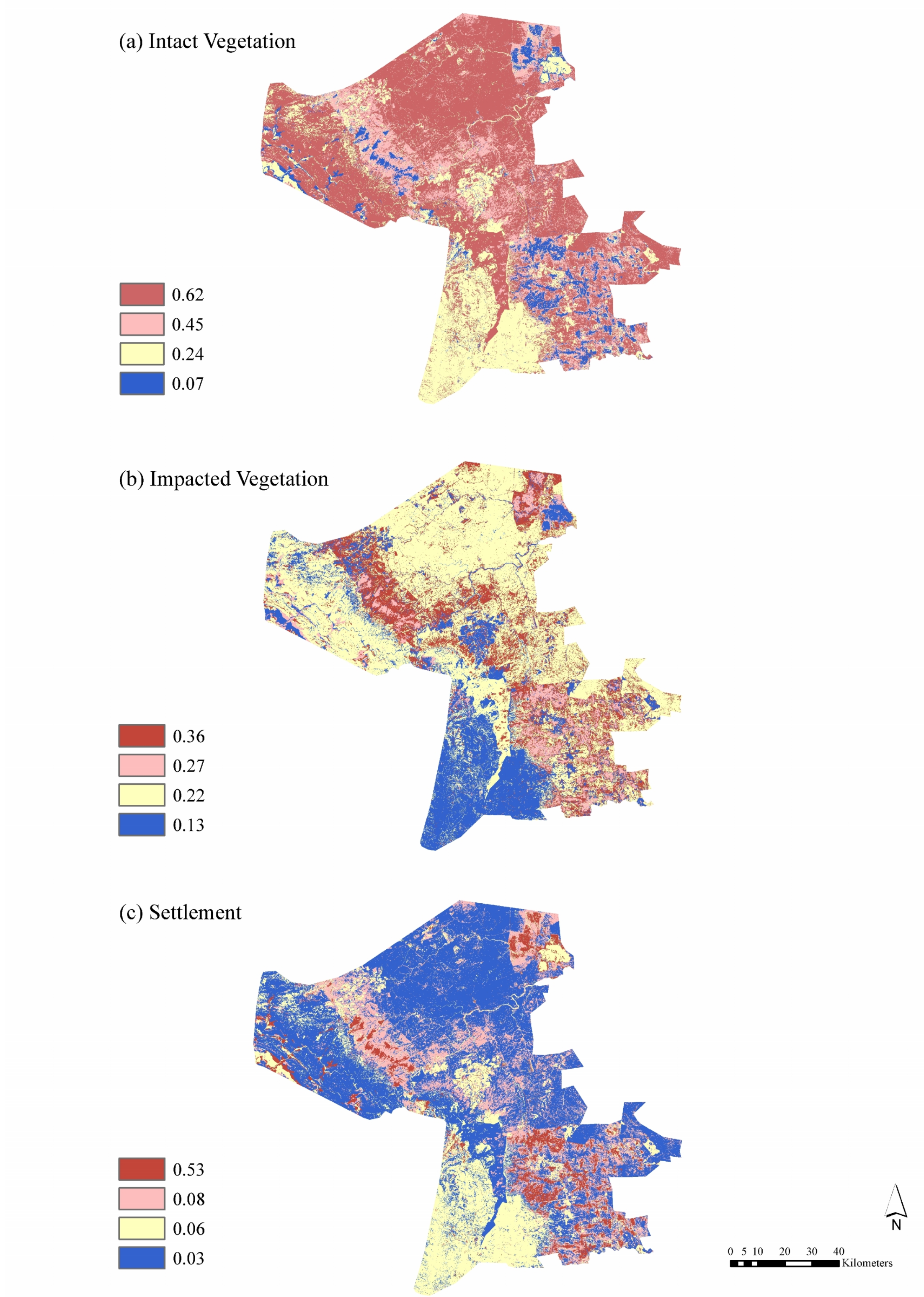
Appendix 2. Markovian transition probability matrix for priority classes: a) 1993 – 2006+*6* and b) 2006 – 2012+*6* simulations

a)

	Cover class				
		2006			
1993		Intact	Impacted	Settlement	Other
	Intact	0.72	0.18	0	0.10
	Impacted	0.26	0.53	0.13	0.07
	Settlement	0.11	0.16	0.61	0.12
	Other	0.23	0.05	0.03	0.69

b)

	Cover class				
		2012			
		Intact	Impacted	Settlement	Other
	Intact	0.62	0.22	0.03	0.13
2006	Impacted	0.46	0.36	0.08	0.11
	Settlement	0.07	0.27	0.53	0.12
	Other	0.24	0.13	0.06	0.56



Appendix 2. Conditional probability surface for each of the priority classes for the simulation period (2006 – 2012), determined on the basis of the Markovian transition probability matrix (Appendix 1). Warmer colours indicate a higher likelihood of persistence into the future

Chapter 6

Conclusions: General Summary, Synthesis and Recommendations for Management and Research

Conclusions: General Summary, Synthesis and Recommendations for Management and Research

6.1 Introduction

In landscape ecology the ultimate goal of characterising landscape change is to determine causality of the change, and make forecasts about future changes with some degree of certainty (Crews and Young, 2013). This in turn develops the support for predictions of resilience across landscapes, as well as providing theoretical generalisations that can be extrapolated to other landscapes where similar circumstances prevail (Crews and Young, 2013). Thus, land-change science has become a critical component of determining sustainability (Wu, 2006; Turner et al., 2007; Termorshuizen and Opdam, 2009), providing the understanding for the biophysical and socio-economic mechanisms of land-use/land-cover change (LULCC; Wrbka et al., 2004; Wu, 2006).

Almost two decades ago Skole (1994) and Lambin (1997) identified priority analyses that might inform land-change science approaches, linking pattern to process and improving understanding of socio-ecological relationships across a landscape. While there have been a number of subsequent advances in methods that capture and quantify LULCC, i.e. remote-sensing technological improvements, as well as those involved in modelling and predicting change dynamics, or in the frameworks developed to uncouple the complexity of socio-ecological systems (SES; e.g. Ostrom, 2009; Ostrom and Cox, 2010), this original approach remains an important interpretation for modern analyses of LULCC.

In this final chapter I summarise the findings of the preceding content chapters, and capture the overlap with Skole (1994) and Lambin (1997)'s guiding principles presented in Chapter 1, i.e. *explanatory analysis*, *spatial analysis*, *temporal analysis*, and the *analysis of the change dynamics* (section 6.2). This chapter synthesises the relevance of the findings for future land-use decision making and further research in the Kruger to Canyons Biosphere Reserve (K2C; sections 6.4 and 6.5), as well as the longer-term sustainability of the BR model for this SES (section 6.3). Beyond this, I have also included a reflection on the evolution of the project (section 6.4), as an organic progression from its initial conception to the current submission.

6.2 Summary of Content Chapters 2 – 5

6.2.1 Chapter 2: Reviewing the BR concept internationally

Chapter 2 reviews the conceptual framework for K2C, presenting the theoretical basis of the BR model and the guiding principles that determine successful implementation in any country. There has been considerable spatial expansion of the Programme globally since first inception in 1974, most recently driven by listings in developing countries where BR's are considered a mechanism for 'conservation to work for the poor'. Yet there are few examples among the 610 sites established by 2012 that had successfully conformed to the MAB Programme's criteria. The reality of aligning conservation and development goals in this model has typically favoured one over the other, with examples of development objectives fuelling broad-scale landscape transformation (e.g. Fu et al., 2004; Ma et al., 2009) and conservation objectives that have ignored socio-economic opportunities (e.g. Price, 2002). However, the review also identified case-studies in which the BR designation in a region precipitated improvements in stakeholder co-operation, and the formulation of a regional identity that united communities (e.g. Bouamrane, 2007; Lange, 2011).

This chapter has emphasised the considerable challenges involved in BR implementation over the last ~ four decades, yet there are potential opportunities inherent in this model that may outweigh these difficulties, especially given the recent revisions to the BR auditing process by UNESCO. In particular, the value of the World Network of Biosphere Reserves (WNBR) to promote inter-disciplinary research endeavours at global scales has been underexploited, with the potential for BRs to act as established 'natural experiments' for sustainability research and our understanding of SESs in general.

Recently however, at the twenty-fifth session of the MAB International Coordinating session in May of 2013, the MAB Secretary introduced new documentation to the WNBR, presenting ideas for a preliminary approach for the future of the WNBR and the MAB Programme at large (2014 – 2021), and likely to address the 'under-exploitation' identified in Chapter 2. This future directive is placed firmly in the context of the Rio+20 conference (United Nations Conference on Sustainable Development, June 2012), and based on an evaluation of the 2008 Madrid Action plan. It focuses specifically on the unrealised opportunities for the WNBR in both national and international research arenas, taking advantage of global positioning of BRs to play key roles at the interface of environmental conservation, research on human-environment interactions, sustainable development, cultural identity, and learning for a sustainable future in emerging international forums (UNESCO, 2013).

6.2.2 Chapters 3 – 5: Understanding the changes in the underlying landscape mosaic

Chapter 3 forms the basis of the *explanatory assessment* of land-cover change in K2C. The choice of land-use options for K2C, i.e. future development expansions / conservation initiatives is likely to be highly contested (e.g. Ferrar and Lötter, 2007), and often, due to the political land-use legacies in the sub-region, also a strongly emotive consideration (e.g. Carruthers, 1995). Chapter 3 identifies the most important historical changes in the BR, identifying the major contributors to spatial change across K2C. While the formal industrial sectors (forestry, agriculture, and mining) contribute substantially spatially to the landscape, landscape change dynamics were driven by the changes in, and spatial interactions between, the Settlement expanse, (human-) Impacted Vegetation and Intact Vegetation cover.

The settlement expanse does not accurately represent human population size in this landscape, and rather is captured as the 'footprint' of settlement space. Chapter 4 indicated that over the study time period (1993 – 2006), population growth alone was not responsible for the expansion of settlement areas, with village sizes often stabilising (see Fisher et al., 2012), yet settlement areas presumably expanding due to changes in household composition (see Madhavan and Schatz, 2007). This is a trend that has become more pronounced in the subregion until 2012, i.e. the 2006 – 2012 observation period. Chapter 5 has presented national and community census data (1996 – 2001 – 2007 – 2011) that depict increasing population growth rates in K2C since 2001, as well as consistently declining average household sizes (Bushbuckridge Local Municipality, 2013).

Chapter 4 presents the s*patial analysis* of land-cover change, assessing the process of change in the Settlement, Impacted Vegetation and Intact Vegetation cover classes through the use of landscape metrics. In this chapter, as well as in the subsequent Chapter 5, these land-cover

classes have been related to the dual management objectives of all BRs globally: development, human-driven resource use and environmental protection.
Chapter 4 quantifies the spatial relationships between these cover classes, illustrating the encroachment of settlement areas into neighbouring rangelands, with the footprint of resource utilisation consolidating rangelands, with consequences for Intact Vegetation. Chapter 4 indicates the spatial sequence of change, identifying that large areas of conversion to Impacted Vegetation require a 'kernel' of existing Impacted Vegetation cover to initiate transformation. This has important consequences for sustainability issues in this SES (e.g. conservation values, resource harvesting) – these smaller losses of Intact Vegetation, (i.e. facilitated through the initial foray for fuelwood harvesting), may act as the as a precursor to the erosion of larger areas, and the resultant degradation of ecological values (e.g. Luoga et al., 2005; Mwavu and Witkowski, 2008; Brady et al., 2009).

Chapter 5 presents the *temporal analysis* of change in K2C, updating the original study time frame with the addition of 2012 data. Chapter 5 presents the changes in the spatial extent, rate and intensity of landscape change, exploring the *change process* across two temporal analysis periods.
The time-series analysis revealed new insight into the process of land-cover change across K2C, information that would have been overlooked had I retained a two-point observation of change. The increased rate of change in the recent 2006 – 2012 observation period poses challenges for landscape management, with future predictions of escalating transformation likely to undermine BR sustainability, and the ability of the zonation to continue functioning as the model intends. Historical landscape trends predict a further 5.1% Intact Vegetation loss in the next 6 years (2012 – 2018), predictions that were informed by the likely spatial sequences of change identified in Chapter 4. While the majority of K2C's Intact Vegetation remains secure within the existing regional network of PAs, the likelihood of further losses of Intact Vegetation from the transition zone of the BR is not improbable. However, based on previous transformation in this zone, the scale of this loss is perhaps unrealistic for a 6 year period. However, over the next 12 years, Chapter 5 illustrates such a loss is possible, and given the spatial extent of such hypothetical loss (> 500km^2), this will necessitate pre-emptive management intervention before it escalates to this point.

6.3 K2C into the Future: Options for Biosphere Reserve Monitoring and Management

The scheduling of future management interventions, e.g. potential transition zone interventions, or issues around access to provincially managed PAs (Chapters 4 and 5), and monitoring of proximity to critical change thresholds (Chapter 5), will necessitate continual reassessment of land-cover changes across the BR. The time lapse between actual change and temporal analysis will affect detection sensitivity and associated management decisions (Huang et al., 2010). However, the creation and validation of land-cover maps is a time-intensive process, requiring expert human interpretation and familiarity with a suite of remote-sensing software. K2C management is a largely voluntary commitment to the BR, held in addition to other responsibilities elsewhere, and the additional effort needed to consistently update and review land-cover from unprocessed imagery, is beyond what may be considered realistic.

Yet, given the importance of broad-scale analysis to contextualise the BR, i.e. establishing linkages between regulatory actions, subsequent land-use and policy decisions (Lunetta et al., 2006), such assessments cannot remain bound to the occasional, and potentially coincidental, research project.
Rather, in this regard, automated land-cover mapping and change detection products may offer considerable value for K2C management in the future.

Over the last five decades, there has been considerable research into automating land-cover classification and change detection across time-series (e.g. Stauffer and McKinney, 1978; Singh, 1989; Knudsen and Olsen, 2003; Im and Jensen, 2005; Kennedy et al., 2007; Stueve et al., 2011), with the majority of these involving complex statistical / machine-learning algorithms and high performance computational capabilities (Coppin et al., 2004; Hansen and Loveland, 2012). Relating usable end-products to natural resource management requires some practical considerations around suitability and utility, with trade-offs between available data (spatial, temporal and spectral considerations; Im and Jensen, 2005) and feasible monitoring (environmental considerations, landscape context and research goals; Kennedy et al., 2009).

In the South African context, the Remote Sensing Research Unit (RSRU) of the Meraka Institute, Council for Scientific and Industrial Research (CSIR), (http://www.csir.co.za/meraka/rsru/) may soon have useable automated products appropriate for research and policy development. This includes time-series analysis MODIS data (spatial resolution: 500m, temporal resolution: daily, 10-daily, from 2000 until present), assessed at regional scales, with the potential to deliver maps of environmental change at these large scales. Salmon et al. (2010) indicate that this automated approach has acceptable levels of mapping accuracy, of >76%. However, given the coarse resolution of the MODIS imagery, there is additional research underway assessing the potential for continuous automated Landsat land-cover products over South Africa (spatial resolution: 30m, temporal resolution: 16 day revisit).

Developed initially for the United States and Alaska, the WELD project (Web-Enabled Landsat Data) provides a continuous surface of Landsat^{ETM} imagery, freely-available, and updated at weekly, monthly, seasonal and annual intervals to capture the temporal variations in earth surface dynamics (Roy et al., 2010a; Roy et al., 2011). From a resource management perspective, the WELD product suite of *Vegetation Continuous Field* (VCF) layers offers considerable opportunities, characterising the percentage tree cover, other vegetation cover (excluding trees), bare ground and probability of water across the Landsat mosaic surface (Hansen et al., 2011; Roy et al., 2011).
The RSRU are currently testing the suitability of this approach for South Africa. Data gaps over Africa prior to 1999, and to a lesser degree, low internet bandwidth speeds limiting downloads of large data volumes (compared to those of Europe and the Americas), may pose a challenge for complete spatial and temporal coverage (Roy et al., 2010b). Yet given the recent successes of land-cover characterisations elsewhere (Hansen et al., 2011), such an automated approach to land-surface classification in South Africa looks promising.

6.4 Evolution of the Project and Research Applications

This research was compelled by a number of case studies that indicated changing land-cover distributions at village and community levels in Bushbuckridge Municipality, with consequences for local-scale ecological values and the sustainability of the socio-ecological system here (e.g.

Dovie et al., 2002; Twine 2005, Giannecchini et al., 2007). This, combined with the recognition that PAs could no longer be managed in isolation from the landscape in which they are found, the (then) recent designation of a 'Biosphere Reserve' site in the subregion, and the need for a landscape approach to ensure enduring biodiversity conservation, identified an important research gap for the subregion, i.e. the understanding of broad-scale land-cover change dynamics.

However, when I began this project, there was no explicit consideration of linking potential results to BR policy. Rather the initial approach was largely coincidental within K2C's boundaries, neglecting to some degree the plethora of literature calling for better integration of science-technical perspectives with real-world decision-making (e.g. Cash and Moser, 2000; Castella et al., 2007; Liu et al., 2008; Armitage et al., 2009; von Winterfeldt, 2013). However, as the project developed, and with the clarity of hindsight, this approach has evolved into a more meaningful discourse and information-sharing with key K2C stakeholders.

The BR concept is poorly-known compared to other conservation designations, (with personal experience indicating that the term is often confused with the *earth's biosphere* rather than as a conservation approach *per se*), and in the context of K2C, largely overlooked in comparison to the world renowned protected areas of the Lowveld subregion (Coetzer et al., 2013a).
There is a scarcity of research specific to the landscape-scale of the BR, a phenomenon that is not only restricted to the South African research environment. Chapters 1 – 5 have included the considerable research on changes in natural resource management at local scales within the BR, yet there are known difficulties in extrapolating these results to larger scales (Castella et al., 2007, also Lovell et al., 2002). Thus, the land-cover research presented in the content chapters, and disseminated through publication (with direct uptake by K2C stakeholders), has satisfied an urgent data need for this subregion, providing the regional / landscape-scale context for BR-related decisions.

Together with K2C authorities I have identified opportunities inherent in my research for informing policy and management priorities across the BR. Those relating directly to the results presented in Chapters 2 - 5, include (*sensu* pers. comm. M. Coetzee, M-T. Uys; K2C Network Coordination Unit):

i) development of a marketing approach for the BR, specifically a logo for the K2C 'brand',

ii) development of a formal process for information-sharing amongst K2C stakeholders, with the first stakeholder workshop to formalise the strategy for *i*) and *ii)* having taken place in November 2013,

iii) contribution to the development of multi-criteria for identifying priority areas for protection or intervention

iv) informing the management and expansion of the protected area network at the landscape level,

v) informing integrated land-use planning and development within, but not restricted to, a Spatial Development Framework for the subregion, and

vi) assessing the collective influence of land-use practices on the BR's overarching objectives and long-term sustainability

Sustainability in the context of K2C goes beyond only an ecological consideration of resilience, and incorporates a consideration of resilience linked to social processes and institutional arrangements (see Armitage et al., 2009).
What has been presented here focuses on a largely bio-physical characterisation of this landscape, and while this can be clearly linked to social and governance considerations, these 'other' components of the BR sustainability need to be explicitly addressed. This calls for an interdisciplinary, collaborative approach to further research in K2C (see section 6.5). A priority would be to formalise the importance of the BR model at provincial and national scales as a mechanism for the regional and integrative management of biodiversity, ecosystem services and human well-being (M. Coetzee, pers. comm.).

The modern definition of the BR stresses the role of research and action learning in order to meet the original sustainable development objectives of the Man and the Biosphere (MAB) Programme (Chapter 2). The notion of formalising the importance of BRs through research has considerable overlap with improving the support that existing BRs receive from the South African national government (e.g. financial, implementation support versus theoretical approval;

Chapters 1 and 2). By aligning the targets of individual BRs more explicitly with the key focal areas of the South African government, i.e. climate change or social development strategies, with BR research projects indicating successful outcomes and/or prototype programmes to be initiated elsewhere, the Programme may become the preferred vehicle for implementation of national policies (Pool-Stanvliet, 2013). It is not that these focus areas for development are not already implicit in the BR designation as it has been applied in the South African context, but rather it may be that these projects / potential benefits need to be more prominently marketed to national government.

6.5 Conclusions and Recommendations for Future Research

The Biosphere Reserve (BR) model is a long-term perspective on landscape management that requires a multi- / interdisciplinary approach to ensure successful implementation. The success of the BR approach is closely tied to understanding the complex adaptive socio-ecological systems (SESs) in the regions where BRs are found, with the underlying land-use mosaic raising challenges for BR management.

For K2C the results presented in this thesis indicate that the spatial distribution of LULCC bears little regard for the theoretical BR zonation and the limitations on 'use' that typifies successful BR implementation (Chapters 3 – 5). I appreciate that this is not the result of the BR presence in this landscape, but rather a systemic condition of landscape change related to social, economic and political factors driving land-use decision-making. Yet, regardless, the scale and rate of future land-cover change, as shown by the historical change trends presented and simulated for the future, undermines the notion of a graduated scale of utilisation across K2C, as well as the goal of maintaining existing ecological values and achieving a connected PA landscape for global change scenarios.

K2C has had substantial developmental / implementation success with pilot socio-economic projects implemented effectively at local sites (Chapter 2). K2C has also received international recognition for the stakeholder engagement / alignment occurring in the subregion as a result of the BR's listing here (Coetzee and Biggs, 2012), and has been identified as a 'flagship' BR for the African continent (M. Coetzee, pers. comm.). Given these successes, as well as the considerable investment in its establishment, it is likely that the BR model here will endure. As a result, a priority would be to ensure that LULCC needs to be more explicitly incorporated into the BR's spatial planning. There is evidence from the K2C Network Coordination Unit that suggests that this is already a practical consideration for the BR (section 6.4).

I have identified a number of additional research gaps in each of the content chapters that would contribute to furthering the understanding of this SES. I put forward the following recommendations for further research in the subregion:

- Chapter 4 suggested that the remnant habitat fragments within the transition zone 'soften' (after Franklin, 1993) the heavily utilised landscape here, facilitating dispersal across K2C and increasing heterogeneity in the areas in which they are found. As a result, Chapter 4 argues that these remnants may have considerable conservation merit.

  In this regard, there are two linked opportunities for further research to quantify this supposition:

  i) Determine to what degree these habitat fragments in the transition zone are functional habitat for biodiversity, and to what degree these facilitate dispersal between the PAs in the BR, i.e. between Kruger National Park – Bushbuckridge Nature Reserve – Motlatse River Canyon Nature Reserve.

  ii) Determine at what levels of utilisation these habitat fragments no longer fulfil this role.

  Previous research in communal rangelands in South Africa has indicated that species differ in their response to land-use practices (e.g. Dzerefos et al., 2003; Fabricus et al., 2003; Botha et al., 2004; Smart et al., 2005; Fisher et al., 2012), and that heterogeneous landscape use mosaics promote gamma diversity across a landscape (Fabricus et al., 2003; Tews et al., 2004).

  The intermediate disturbance hypothesis (IDH, after Connell, 1978) predicts that species diversity is highest at intermediate levels of disturbance, and lower at under very frequent / infrequent disturbance regimes. The literature contains considerable debate on the continued validly of the IDH for ecological systems (see recently: Fox, 2013 versus Sheil and Burlem, 2013). Yet I suggest that IDH may signify an appropriate theoretical basis for assessing the links between intensity of resource use and the ability of these fragments to continue to contribute to landscape connectivity and heterogeneity in K2C.

- Throughout this thesis I have referred to the challenges of up-scaling / down-scaling the socio-ecological and economic determinants of land-cover change, due to issues of site specificity and scale dependency (i.e. Crews and Young, 2013). Thus, a priority for research in this SES is to empirically test the effect of scale on the drivers and patterns of change, replicated in different areas of this landscape.
  Remote-sensing technology, combined with extensive fieldwork campaigns that include household surveys, will be instrumental in this regard.

  There are a number of studies that have considered land-cover changes at different scales in this SES (Table 1 include recent relevant examples). However, there exists a research gap in quantifying and linking these results vertically through scale-space, explicitly testing whether the trends captured at one level / resolution in the scale-hierarchy are apparent in the next (e.g. Wheatley and Johnson, 2009). Addressing this issue will address issues linked to causality of change patterns, and allow for the extrapolation of findings beyond these study localities.

  For K2C specifically, given the importance of fuelwood harvesting for local livelihoods in this region, with fuelwood contributing 44% to household harvests (Dovie et al., 2002), I suggest that the links between harvesting for fuelwood, and observed landscape change at multiple scales should take precedence.

Table 1. Recent studies of landscape change undertaken in the Kruger to Canyons Biosphere Reserve, with the scale of observation indicated

Authors	**Minimum mapping unit**	**Scale of study**
Matsika (2013)	Household	Individual villages
Saunders (in prep)	Aerial photography: <3m	Multiple rangelands linked to multiple villages
Fisher et al. (2012) Fisher et al. (in press)	LIDAR (Light Detection and Ranging) imagery: <2m	Management units: Communal rangeland versus Protected Area
Coetzer et al., 2010; 2013b	Landsat imagery : 30m	Landscape-scale: Biosphere Reserve

- The feasibility of alternative and complementary approaches to biodiversity conservation needs to be assessed for K2C, i.e. 'off-reserve conservation' such as stewardship programmes.

Given the extensive spatial investment to formal conservation in the BR, it is unlikely that the establishment of 'new' traditional PAs will offer a complete solution to rectify historical *ad hoc* protected area placement. As a result, agreements between conservation agencies and private / community land-owners are being identified as a mechanism to strengthen the existing PA network and facilitate landscape-scale conservation throughout South Africa (e.g. Pence et al, 2003; Jackelman et al., 2008).

Chapter 5 has presented one such agreement between the privately-owned Sabi Sands Wildtuin and the Amashangane community. Chapter 2 has reported on a local-scale initiative, a community-based natural resource management approach that focuses specifically on medicinal health practitioners and the conservation of scarce medicinal resources in the BR.
The feasibility of additional initiatives needs to be investigated, identifying conservation alternatives appropriate for the subregion, e.g. conservation easements, management agreements and financial incentives (for review Pence et al., 2003). The extensive

stakeholder engagement that has become typical of K2C would play a central role in this regard.

Similarly, there is a requirement for a formal decision-support framework assessing options for ‘off-reserve’ versus ‘on-reserve’ conservation in this landscape (e.g. Pence et al., 2003), assessing the trade-offs between levels of protection, future landscape changes, and the strength of stakeholder relationships.

- In Chapter 5 I have proposed that a further 5% total loss of intact habitat over the next 12 years may represent an important ‘threshold’ of change for K2C. Based on historical trends of landscape change (Chapters 4 and 5, also section 6.2), this does not appear improbable for the unprotected areas, i.e. the transition zone vegetation.

  I have referred to this as an ‘activity switch’ of change for Intact Vegetation, which is based on the formulation of Aldwaik and Pontius (2012). This is a hypothetical threshold, and whether it represents an important ecological change is not known. Future research that facilitates this conceptual-functional linkage is likely to offer considerable benefits for future environmental management, and should be prioritised.

- This chapter has emphasised that sustainability in the context of SESs goes beyond environmental sustainability, but requires a holistic, interdisciplinary approach that includes social, economic and institutional considerations. Chapter 2 argues that BRs, given their prescription for co-management and co-operation between multiple stakeholders, as well as the opportunities they offer as ‘learning sites for interdisciplinary research’ will contribute to studies of sustainability internationally. This is a research opportunity that has not been successfully exploited in the history of the MAB Programme.

  The framework presented by Ostrom (2007; 2009) may be effective in facilitating research that is translatable across BR sites at global scales: a multi-tiered, nested approach to organise research in SESs. The diagnostic framework is intended to identify relevant variables and their subcomponents in a particular SES, and guide the design of research projects and the analysis of findings with regard to sustainability of the system (Ostrom, 2009; Ostrom and Cox, 2010). Empirical testing of the framework is in its infancy (Ostrom, 2009), but Fisher (2013) has interpreted it conceptually for fine-scale sites within K2C with some success.

- All analyses presented in this thesis have excluded Kruger National Park (KNP) from the Biosphere Reserve region, yet central KNP forms a substantial portion of K2C. Thus, there is considerable opportunity for the approaches outlined in Chapters 3 – 5 to be applied to this PA, assessing changes in vegetation cover, and providing further insight to the processes of change, and the landscape effects of these changes across the Park, e.g. climate change, the effect of herbivores or specific management strategies.

  While the drivers of change in KNP differ from those in the matrix areas beyond the Park (e.g. Smith et al., 2013; Wigley et al., 2014), applying the approaches I have presented in the preceding chapters to specific habitat types / broad-scale vegetation units, particularly those of Chapter 5, may assist with *i*) the development of monitoring protocols and measuring the 'performance' of the PA (e.g. McGeoch et al., 2011), *ii*) help identify 'natural' thresholds of change, and *iii*) determine the influence of the K2C matrix area, and the land-cover changes therein, on the functioning of KNP long-term. These additional analyses will assist with the sustainability considerations (i.e. Chapter 1, Objective 3) of the Biosphere Reserve approach for this renowned conservation landscape.

6.6 References

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