A pluralistic, socio-ecological approach to understand the long-term impact of mountain conservation

A counterfactual and place-based assessment of social, ecological and hydrological change in the Groot Winterhoek Mountains of the Cape Floristic Region

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DECLARATION

I, Petra Brigitte Holden, hereby declare that the dissertation for the degree of Doctor of Philosophy is my own work and that it has not previously been submitted for assessment or completion of any postgraduate qualification to another University or for another qualification.

Signature Signed by candidate Date 30 November 2017

ABSTRACT

The problem: For protected areas to remain relevant, we need to understand their impact on a wide set of conservation objectives and environmental outcomes. We also need to evaluate how this influence relates to the socio-ecological environment within which they occur. This is a complex endeavour requiring a pluralistic approach, which draws on a wide range of interdisciplinary fields.

Research question: This thesis addresses the following question: What effects do mountain protected areas have on ecosystem services over time and how does this influence relate to broader socio-economic and ecological drivers of landscape change?

Aim and objectives: I use a pluralistic, socio-ecological framing to assess the impact of ~40 years of mountain protection, drawing on comparisons of ~30 and ~40 years before and after protection respectively, with an adjacent area of similar terrain informing scenarios of counterfactual conditions. I also investigate what types of values (economic and intrinsic) are important when determining the impact of mountain protected areas.

Thesis approach and methods: I operationalise the concepts of socio-ecological systems, ecosystem services, land use transitions and counterfactuals to investigate socio-ecological change and how it relates to protected area impact in the Groot Winterhoek, a mountain catchment in the south-western Cape of South Africa. This mountain catchment is important for regional water supplies for agricultural and domestic uses and falls in the Cape Floristic Region, a global biodiversity hotspot. It is comprised of privately owned mountain wildlands and a wilderness-protected area, known as the Groot Winterhoek Wilderness Area, established in 1978 (gazetted in 1985) which forms part of the Cape Floristic Region World Heritage Site.

I combine methods from social science, ecology, environmental geography, geomatics and hydrology to understand the history of land use and cover (land use/cover) and associated ecosystem service trade-offs, how they are perceived by landowners as well as their wider impact on the region. Specifically, I assess the impact of protection on land use/cover, vegetation, fire and water flows over the last ~50 years, by comparing and contextualising results of change within the protected area to alternative scenarios of "no protection" (the counterfactual conditions).

Vegetation and land use/cover change inside the protected area were determined respectively using 72 repeat terrestrial photographs and vegetation surveys, and an analysis of orthorectified aerial imagery. Methods used to construct the counterfactual scenarios of mechanisms (e.g. changes in land use/cover) that would likely drive vegetation changes inside the protected area included: i) 60 repeat surveys and in-depth interviews with landowners adjacent or proximal to the protected area owning unprotected land of similar terrain to the protected area; and ii) land use/cover change analysis of orthorectified aerial imagery of adjacent unprotected land of similar terrain before and after protected area establishment.

This latter information was used to understand the role of the protected area in driving vegetation changes inside the protected area.

Social, biophysical and remote sensing results were directly used to parameterise land use/cover components of a hydrological model to determine the influence of protection on water flows. Specifically, water flows were simulated for the current state of the environment inside the protected area as well as for several counterfactual scenarios i.e. the alternative land use/cover scenarios of "no protection". These counterfactual scenarios included land use/cover at two-time steps of ~30 and ~8 years before protection and one-time step ~40 years after protection both inside and outside the protected area.

Results:

Long-term change in ecosystem service use outside the protected area on privately owned land of similar terrain to inside the protected area (Section 3):

Over the last ~50 years, outside the protected area, there was a shift from livestock-based, subsistence agriculture and small-scale farming to a diversified set of ecosystem service uses. The combined area of grazing and wildflower harvesting declined by 39%, while the number of landowners using the mountains for personal nature-based recreation and ecotourism increased by 61% and 23% respectively. Agriculture intensified in suitable areas of mountain land with the number of landowners cultivating land increasing by 20%. Exogenous socio-economic drivers associated with globalisation and economic growth were important causal mechanisms of land use change. Landowners valued mountain protection for intrinsic and non-use reasons (73-80% of landowners), including existence, bequest and option values, as well as for the indirect use of water supply (72% of landowners) in comparison with direct use reasons such as spiritual/cultural experiences and nature-based recreation inside the protected area (18 and 50% of landowners respectively). Personal, nature-based recreation outside the wilderness-protected area was associated with valuing the protection of mountain land for intrinsic and non-use reasons.

Long-term vegetation change inside the protected area and plausible mechanisms driving vegetation change (Section 4):

Inside the mountain protected area, fynbos vegetation cover increased on average between 11 and 30% and there were significant declines in bare ground and rock cover. Increases in fynbos vegetation were comprised mostly of shrubs on shale-band areas, restioids-sedges on hydromorphic sandstone-quartzitic sites and a mixture of growth form types on partly shale-derived soils. Significant declines in large adult proteoids were also observed in some areas. There were no clear trends in thicket-forest canopy and basal vegetation cover for rocky outcrops, despite certain sites showing declines in thicket-forest canopies and increases in pioneer species around thicket-forest margins. Increases in vegetation cover occurred despite reduced summer rainfall and annual wind run and increased annual temperatures over the last ~40 years. Therefore, positive changes in fynbos vegetation were likely achieved due to the elimination of livestock-based subsistence agriculture and small-scale farming practices of the past. These changes have resulted in an increase in fire return intervals, fuel accumulation and fire intensities. However, these latter changes in land use/cover also occurred outside the protected area (see results summarised for Section 3 above and Section 4 below) and therefore cannot be attributed to protected area establishment.

Land use/cover and the influence on water flows inside the protected area compared to counterfactual scenarios of no protection (Section 5):

Declines in grazing and changes to the fire regimes occurred regardless of the protected area boundaries. In the past, there was a high frequency of small, low intensity fires across the landscape, both inside and outside the protected area. More recently, fires have been actively suppressed and this results in the build-up of biomass and the development of extensive, high intensity fires which, under suitable conditions, burn large expanses of the mountain catchment. Hydrological modelling showed that a high intensity burning regime negatively affected streamflow regardless of protected area boundaries. Streamflow increased by more than 80% under high flow conditions and decreased by more than 40% under low flow conditions relative to an unburnt 'natural' scenario.

Over the last 50 years there has also been a substantial increase in dams, buildings and roads and minor increases in cultivation outside the protected area. This has been avoided inside the protected area where these land use/cover classes declined. If the increase in these land use/cover types observed outside the protected area occurred inside the protected area this would have resulted in reductions in daily streamflow leaving the protected portion of the catchment. For example, outside the protected area reductions of 8% to 25% of streamflow were observed during mid and low flow conditions respectively, particularly during dry years, in comparison to a 'natural' scenario. In contrast, inside the protected area streamflow recovered from past conditions to more closely resemble the natural flow conditions of the catchment.

Therefore, had the protected area not been established there would have been losses in streamflow from the catchment as well as an increase in the degree of fragmentation within this mountain area. However, with increased water storage and fragmentation outside the protected area has also come increased socio-economic opportunities such as employment and local opportunities for ecotourism and sustainable agriculture e.g. indigenous cut flows. This highlights the importance of maintaining various forms of land management systems (multifunctional landscapes) within mountain ecosystems but also the need to understand the sustainability of different land management system types. Determining appropriate land management systems for mountain areas should be based on a full understanding of the impacts on ecosystem service benefits and costs at local and regional levels between social groups both spatially and temporally.

Broader significance: This thesis contributes to the conservation literature on two main fronts. Firstly, it contributes conceptually and theoretically to understanding the dynamics of ecosystem services in relation to mountain protection. Secondly, it contributes methodologically by using an inclusive, trans- and interdisciplinary research approach for evidence-based conservation at a place-based and landscape level. The study provides a case study example of the positive impact that mountain protection has on water-related ecosystem services, notably by maintaining streamflow throughout high to low flow periods and during dry years. It also provides clear evidence that ecosystem service trade-offs do not remain constant over time and shows that intrinsic and non-use values are required when describing the importance of mountain protected areas.

In terms of understanding the impact that protected areas have in mountain regions, the research shows that complex processes are at play that extend beyond the boundaries of a specific protected area in both time and space. Interactions between global and local drivers were found to be prominent causal mechanisms of socio-ecological change and ultimately determined the influence of mountain-protection on land use/cover, fire, vegetation and water-related ecosystem services. The thesis emphasises that counterfactual framings are necessary to understand and attribute the impacts of protected areas on environmental outcomes, however pluralism and socio-ecological approaches are critical to determine plausible counterfactual conditions.

This thesis focused only on landowners adjacent and proximal to the protected area owning the majority of mountain catchment land of similar terrain. It is likely that multiple socio-economic trade-offs have occurred between different social groups and generations at both local and regional levels. Understanding how the disadvantages and benefits of the impacts of protected areas are apportioned across the landscape and temporally is an aspect that requires future research. Central to this would be to fully consider how human well-being is influenced both upstream and downstream, including at regional levels, and between social groups and across generations. Considering the impact of protected areas on the full range of ecosystem services and linking this to societal preferences and perceptions should be incorporated into the overall goal of developing an evidence base for conservation. This is because it is both scientific evidence and societal change that can determine protected area persistence and thus long-term protected area impact.

Keywords: counterfactuals, ecosystem services, land use transitions, hydrological modelling, land use/land cover, mixed methods, pluralism, protected area impact; socio-ecological systems, trans- and interdisciplinarity

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Other: Maps throughout this thesis were created using ArcGIS® software by ESRI (ESRI 2015). Plots throughout this article were generated in Rstudio using ggplot2 (Wickham 2009; R Core Team 2016).

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Section 1. Introduction

Over the last century, protected areas have remained a core strategy for conservation. However, given an uncertain global future, for protected areas to remain relevant and thereby effective, an understanding of their socio-ecological context is required (Cumming et al. 2015; Cumming 2016). The environment can be influenced by a multitude of variables regardless of the presence or absence of a protected area. Therefore, there is a need for reliable evidence about the degree to which protected areas influence environmental outcomes while accounting for global and local socio-ecological circumstances (Miteva et al. 2012). This includes acquiring an understanding of the socio-ecological trajectories that have resulted in protection as a strategy and how trajectories of socio-ecological change mask or mimic protected area impacts (Ferraro & Pressey 2015; Pressey et al. 2015).

Following the logic presented by Pressey et al. (2015), the basic purpose of protected areas is to avoid loss of ecosystems, species or other valued aspects of the natural environment such as ecosystem services. Therefore, protected areas should be measured in terms of how much loss has been avoided. This is referred to as impact. To determine protected area impact, an attempt must be made to estimate (quantitatively and/or qualitatively) what the conditions would have been like in the absence of a protected area. These are referred to as the relevant counterfactual conditions (Pressey et al. 2015). If the proposed counterfactual outcome is considered much worse for conservation, then protection has had a large impact (Ferraro & Pattanayak 2006; Ferraro & Hanauer 2014a; Ferraro & Pressey 2015; Pressey et al. 2015). The use of outcome-focused protected area impact evaluations has rarely been implemented, although it is being increasingly drawn on in the global literature (Craigie et al. 2015; Ferraro & Hanauer 2015; Ferraro & Pressey 2015; Hanauer & Canavire-Bacarreza 2015; Pfaff et al. 2015). The impact evaluation field recognises the importance of before-after and with-without comparisons. It adds to this by drawing on scientific advances made in the ability to infer causality from non-experimental data (Ferraro & Pressey 2015).

The counterfactual approach of protected area impact evaluations has much to contribute to the field of conservation biology. However, under the current framings of evidence-based conservation, in the protected area impact evaluation field, preference is often given to certain types of knowledge, methods and information. In particular, there is an emphasis on quantitative methods (Bennett 2016). In many cases, it is also assumed that qualitative data and information sources are of limited value for the development of credible evidence on protected area impacts. Therefore, most protected area impact studies that follow counterfactual approaches use mainstream remote sensing methodologies and do not consider a full range of qualitative and quantitative data sources and analyses. As a result, changes in multiple land use and cover types and long-term changes in ecosystem services have not been adequately considered in the protected area effectiveness literature that relies on counterfactual framings. This is largely because the use of trans- and interdisciplinary approaches and in particular the integration of social sciences into protected area impact evaluations has been limited (Naughton-Treves et al. 2005; Fox et al. 2006; Miller et al. 2008; Roy et al. 2013; Bennett et al. 2017).

Sustainability science (Kates 2011; Shahadu 2016) provides the foundation to achieve epistemological and methodological pluralism (referred to as pluralism) with a focus on achieving trans- and interdisciplinary methodologies in research. Pluralism recognizes that, in any given research context, there may be several valuable ways of knowing and doing (i.e. various forms of methods and knowledge), and that accepting this can lead to a more successful, integrated study (Isgren et al. 2017). Numerous operational and institutional barriers exist to achieving pluralism in studying the impact of protected areas (Fox et al. 2006; Roy et al. 2013; Fischer et al. 2014; Pooley et al. 2014), and the importance of integrated knowledge and diverse approaches is often overshadowed by calls for generalisations grounded in credible evidence (Bennett 2016).

Studies are usually expected to have “elegant” research designs, clear *a priori* hypotheses, “strong” sampling approaches, and robust statistical analyses and interpretation (Laurance et al. 2012). Thus, there is a focus on large sample sizes, increased spatial coverages and, in particular, quantitative data analyses (Fischer et al. 2014). This is further complicated by the difficulty of being able to grasp the field of sustainability science in its entirety, a field that is also grappling with its own aspirations of achieving pluralism in research approaches (Shahadu 2016). Tensions exist between the need for pluralism and flexibility on the one hand and the requirement for precision in methodological approaches on the other (Isgren et al. 2017).

In this thesis, I draw on the foundational goals of sustainability science associated with pluralism and the concept of counterfactuals to study the impact of mountain protection on vegetation, land use and cover (land use/cover) and water flows within a socio-ecological context and at a place-based and landscape level. I use an inductive and iterative approach to develop the overall thesis methodology. This includes building on the progress made in the protected area impact evaluation literature, and in particular the use of counterfactuals, but aligning this with advances made in the fields of sustainability science, conservation biology and land change science on trans- and interdisciplinary methods for understanding change in socio-ecological systems. The central link is the concept of counterfactuals and the use of many methods and approaches to develop an understanding of these counterfactual conditions for the protected area under study. Plausible counterfactual conditions are developed, using a range of methodologies and knowledge sources, which are used to describe socio-ecological change and drivers of this change. The counterfactual conditions are then used to contextualise change inside the protected area and thus validate protected area impact.

My case study area is the Groot Winterhoek Mountains in the Cape Floristic Region, a global biodiversity hotspot and World Heritage Site in South Africa. The Groot Winterhoek is a national strategic water source area, which supplies the Berg-Olifants Water Management

Area, and is critically important for the Western Cape Water Supply System but also for certain local downstream agricultural, economic and domestic activities.

The Cape Floristic Region was identified for the study due to the wealth of knowledge and research available for the region. This includes literature on spatial prioritisation for conservation (Cowling et al. 2003a; Pressey et al. 2003), integrated socio-ecological conservation assessment and planning (Pierce et al. 2005; Knight et al. 2006; Egoh et al. 2007, 2008; Pasquini et al. 2010; Rouget et al. 2014), environmental history (Grove 1987; Bennett & Kruger 2015; van Wilgen et al. 2016a), patterns of diversity (Cowling & Proches 2005; Bergh et al. 2014), the influence of land cover change (notably afforestation and fire) in mountain catchments on water flows (Wicht 1971; Van Wilgen 1981; Bosch & Hewlett 1982; van Wilgen et al. 1992, 2016a; Scott 1993, 1997; Dye 1996; Le Maitre et al. 2014), the influence, including success, of management interventions related to alien plant clearing (Van Wilgen et al. 1996; Currie et al. 2009; van Wilgen 2009; McConnachie et al. 2015, 2016a, 2016b; Kraaij et al. 2017) and restoration programmes in mountain catchments (Fill et al. 2017), the biology and ecology of the region (Cowling et al. 2003a; Allsopp et al. 2014) and the threats facing the region (Midgley et al. 2003; Rouget et al. 2003b, 2014).

In addition, despite challenges associated with achieving pluralism in studying protected area impacts, South African conservation scientists working in the Cape Floristic Region have made substantial progress towards achieving trans- and interdisciplinary methods and bridging the gap between conservation assessment, planning and implementation by adopting social science theory and methodology and transdisciplinary processes (Cowling et al. 2003b, 2010; Lochner et al. 2003; Younge & Fowkes 2003; Pierce et al. 2005; Knight et al. 2006; Knight & Cowling 2007; Le Maitre et al. 2007; Pasquini et al. 2010; Reyers et al. 2010). Conservation scientists in the Cape Floristic Region have also made advances in mainstreaming spatial biodiversity assessment products into conservation and land use decision making (Reyers et al. 2007) and mainstreaming the concept of ecosystem services into land use planning (Cowling et al. 2008).

Mountains are the focus of the thesis due to the limited research that uses a counterfactual framing to understand the impact of mountain protected areas on multiple land use/cover types and water-related ecosystem services. For example, despite wide recognition of the importance of protecting mountain areas for sustaining water flows (Viviroli et al. 2007, 2011; Garrido & Dinar 2009), there is a dismissal of the impact of mountain protection in achieving conservation goals in the protected area impact evaluation literature (i.e. the literature that uses counterfactual approaches) (Joppa & Pfaff 2009; Palomo et al. 2014; Pressey et al. 2015). This is because in many cases, due to the low extractive use potential of mountain environments, the counterfactual conditions end up being similar to the protected area state. Therefore, although mountain areas are considered critically important for the conservation of biodiversity and ecosystem services, in particular water-related ecosystem services, this is often overshadowed in protected area impact evaluations that use counterfactual approaches (Joppa & Pfaff 2009) due the course level of analyses and the limited engagement with pluralistic approaches in their methodology (Naughton-Treves et al. 2005; Fox et al. 2006; Miller et al. 2008; Roy et al. 2013; Bennett et al. 2017).

The introduction of this thesis provides an overview of the context of conservation in the Cape Floristic Region. This is then contrasted with existing global trends in the protected area impact evaluation literature. The importance of counterfactual framings and socio-ecological context for protected area impact evaluations is discussed and linked to place-based and landscape level analyses. I review existing frameworks and concepts that have been used to diagnose and evaluate socio-ecological conditions including resilience thinking and theory (Folke et al. 2010; Folke 2016), the Social-Ecological Systems Framework (Ostrom 2009), and the concepts of ecosystem services (MEA 2005; Costanza et al. 2017), land use transitions (Foley et al. 2005; Lambin & Meyfroidt 2010) and socio-ecological systems (Folke & Berkes 1998; Walker & Salt 2012; Fischer et al. 2015). I argue for the usefulness of pluralism, as a boundary concept, for promoting trans- and interdisciplinarity including the integration of multiple concepts for assessing the impact of protected areas on environmental outcomes while using a counterfactual framing at a place-base or landscape level. I conclude the introductory section of the thesis with an overview of the research aim and objectives and a brief outline of the thesis.

1.1 Conservation in the Cape Floristic Region

Substantial efforts have been made to conserve the Cape Floristic Region over the last century. Most conservation interventions, especially earlier efforts, have been associated with protected area establishment and expansion. For example, by 1900, less than 1% of the Cape Floristic Region was protected; this grew to 10% in 1945. Expansion was further increased during the late 1960s and early 1980s. By 1990, protected areas covered 20% of the Cape Floristic Region and grew to just over 21% in 2000 (Rouget et al. 2014). The most vigorous establishment and expansion of the protected area estate in the Cape Floristic Region was driven by two pressing concerns during the 20th century. The first concern was to reduce the destruction of indigenous forests and the second was to protect mountain catchments and their associated water supply from the negative effects of private land use activities (Rouget et al. 2014; van Wilgen et al. 2016a). As a direct result, many of the current protected areas are biased towards mountainous landscapes. Mountains are therefore well represented in protected area targets in comparison to the lowlands (Rouget et al. 2003a; Department of Environmental Affairs 2016; van Wilgen et al. 2016a).

Similar to global trends (Cumming 2016), statutory and non-statutory protected areas are still used and seen as the most important tool for conservation in the Cape Floristic Region. For example, since 2000, there has been a 7% increase in protected area coverage resulting in more than a quarter of the Cape Floristic Region being protected under various agreements. Formal protection, underpinned by strong legislation, represents 70% of this area (Rouget et al. 2014; van Wilgen et al. 2016a). Over the last 15 years, however, there has been a shift in this approach to conservation largely because the lowlands are requiring substantial inputs for safeguarding biodiversity. The aim has been to establish and expand protected areas to conserve critical biodiversity areas in the lowlands and to build institutional capacity and mainstream biodiversity conservation in relevant government sectors, industry and business (Rouget et al. 2014; van Wilgen et al. 2016a). In the uplands, major advances have been made in understanding the hydrological impact of alien plants on water supply and the political decisions that need to be made to mitigate these impacts (van Wilgen et al. 2012, 2016a; Fill et al. 2017). The results of this work have influenced both policy (e.g. the Water Act of 1998 and the National Environmental Management Biodiversity Act of 2003), and practice (e.g. the Working for Water Programme) (van Wilgen & Wannenburgh 2016).

The conservation history of the Cape Floristic Region has been well researched through a series of environmental historical studies and monologues (Grove 1987; Pooley 2012, 2015; Bennett & Kruger 2015; van Wilgen et al. 2016a). The biology and ecology has also been relatively well studied (Allsopp et al. 2014) and there are numerous studies focused on conservation planning and sampling outcomes (i.e. the extent to which aspects of biodiversity are represented within protected areas (Pressey et al. 2015)) for the region (Rouget et al. 2014). Conservation scientists in the Cape Floristic Region have been leaders in the fields of conservation assessment and planning. Conservation assessment in the region has involved identifying spatial priorities for conservation action considering biodiversity, socio-economic and ecosystem service aspects (Cowling et al. 2003a, 2003b; Lombard et al. 2003; Pressey et al. 2003; Younge & Fowkes 2003; Egoh et al. 2007; Knight & Cowling 2007). Conservation planning has also involved significant engagement with stakeholders in order to develop implementation strategies for the results of conservation assessments (Lochner et al. 2003; Knight et al. 2006; Egoh et al. 2007; Cowling et al. 2010; Pasquini et al. 2010). Much progress has been made in terms of integrating ecosystem services into conservation assessments and planning as well as using trans- and interdisciplinary approaches to conservation assessments and planning with a number of approaches and frameworks being proposed in this regard (Pierce et al. 2005; Egoh et al. 2007, 2008; Le Maitre et al. 2007; Reyers et al. 2007, 2010; Cowling et al. 2008).

Despite the extensive protected area estate and substantial conservation assessment and planning literature and community of practice in the Cape Floristic Region, using a counterfactual framing to understand the long-term impact of protected area establishment and expansion on multiple land use/cover types has been limited. Specifically, the impact of protected area establishment on previously, privately utilised or unmanaged mountain land has not been investigated following protected area impact evaluation methodologies i.e. using the concept of counterfactuals (Rouget et al. 2014). In many cases, protected areas were established to reduce the frequency of burning associated with subsistence farming and to remove completely the incidence of grazing, especially by goats and horses on mountain fynbos. Protected areas were also established to remove the threat of small-scale cultivation on mountain land in order to protect vegetation cover, reduce soil loss and ensure the continued supply of clean water (van Wilgen et al. 2016a).

Given that mountainous regions in the Cape Floristic Region were the priority for conservationists in the past, they provide ideal case studies for understanding the effect of protected area establishment on specific environmental outcomes. This thesis builds on the existing trans- and interdisipclinary approaches used in the Cape Floristic Region for conservation assessment and planning but aligns this with the literature on protected area impact evaluations and in particular the use of counterfactuals. The thesis aims to build on the progess made by conservation scientists in the Cape Floristic Region to align conservation assessment and planning with sustainability science, in particular the integration of ecosystem services into conservation assessment and planning methodologies. Trans- and interdisicplinary approaches are currently lacking in international approaches to assessing protected area impact using counterfactual methodologies (Naughton-Treves et al. 2005; Fox et al. 2006; Miller et al. 2008; Roy et al. 2013; Bennett et al. 2017).

1.2 Counterfactuals and measuring protected area impact

Central to the study of protected area effectiveness, within an emerging field known as protected area impact evaluation, is the concept of counterfactuals (Ferraro & Pattanayak 2006; Ferraro 2009; Miteva et al. 2012; Ferraro & Hanauer 2014a, 2014b, 2015; Pressey et al. 2015; Ferraro & Pressey 2015; Hanauer & Canavire-Bacarreza 2015; Jones & Lewis 2015; Coetzee 2017). Counterfactuals are alternatives to past events, actions or states. They represent scenarios, or thoughts of what might have been or what might have occurred under a different set of circumstances (Epstude & Roese 2008). Conservation benefit is simply the difference between environmental outcomes with conservation action and without (Maron et al. 2013). The concept is applied to evaluate what conditions would have been like in the absence of protected areas i.e. the relevant counterfactual conditions. Therefore, the aim is to determine how much loss of habitat cover, ecosystems, species and/or ecosystem services has been avoided within the area that is protected (Ferraro & Pattanayak 2006; Ferraro 2009; Ferraro & Pressey 2015; Pressey et al. 2015).

The primary mechanisms through which protected areas can affect the environment is by influencing human decisions to use or consume resources in areas of significant biodiversity and ecosystem services (Ferraro & Hanauer 2015). However, multiple confounding factors influence land use decisions. A recent framework has been suggested by Ferraro & Hanauer (2015), drawing on a typical causal inference diagram known as a causal directed acyclic diagram, to understand the influence of confounding factors on protected area impact and for determining appropriate counterfactuals (Figure 1.1). In this framework, the treatment is the form of protection assigned to an area. Mechanisms are agents that influence the causal pathway between the treatment and the outcome. Specifically, mechanisms are the threats to ecosystem services that can be affected by the protected area. These include infrastructure, such as roads or buildings, and land use such as cultivation or grazing. Moderators modify the strength of the effect of mechanisms along the causal pathway between treatments and outcomes. Moderators, such as climate, however, are not affected by protection. Confounding variables, such as market forces, jointly affect treatment, mechanisms and outcomes, and therefore may mimic or mask the impacts of protection. For example, confounding variables can result in a decline in a specific land use type or a switch in land use regardless of conservation efforts. The impact of a protected area is the value added to a counterfactual estimate of the outcome. In other words, the difference between with and without protection is the impact and is defined as the outcomes arising from protection relative to the counterfactual of no protection (Ferraro & Hanauer 2015; Pressey et al. 2015; Coetzee 2017).

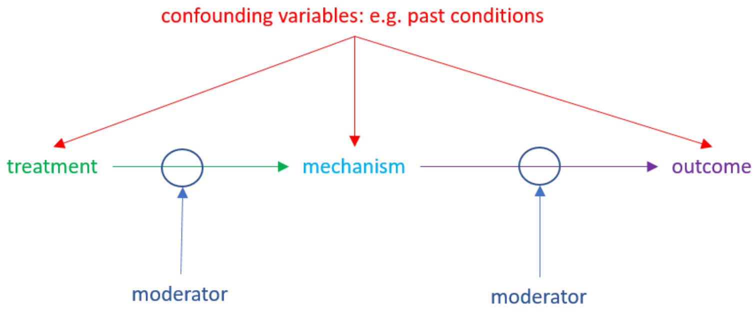


Figure 1.1 Recent framework suggested by Ferraro & Hanauer (2015) for distintangling the effects of mechanisms, and moderators on protected area impact.

Using the counterfactual framing, conservationists have increasingly taken to using the abundance of remote sensing imagery and products that have become available over the last 30 years. This has been driven in the quest to develop scientific evidence for protected area effects on land use and cover (land use/cover). The trend has been progressively growing due to increased access to regional scale remote sensing products, which provide countless pixels to increase sample sizes and spatial scales, and from which covariates can be extracted for matching statistics and regression analyses (Naughton-Treves et al. 2005; Nelson & Chomitz 2011; Jones & Lewis 2015). As a result, numerous studies exist which have compared conditions inside with those outside protected area networks (Andam et al. 2008; Pfaff et al. 2009, 2014; Gaveau et al. 2009, 2009; Joppa & Pfaff 2011; Nelson & Chomitz 2011; Beresford et al. 2013; Sieber et al. 2013, 2013; Ferraro et al. 2013; Vergara-Asenjo & Potvin 2014; Carranza et al. 2014; Haruna et al. 2014; Bowker et al. 2017).

This increasing body of literature has proved valuable for creating generalisations about the impact of protected areas (Miteva et al. 2012; Geldmann et al. 2013). Studies, however, often use post-protection measures without accounting for pre-protection measures of confounding factors, which ultimately can influence post-protection results (Ferraro & Hanauer 2015). Furthermore, change in covariate values used in matching and regression analyses are not considered (Jones & Lewis 2015). When studies do consider pre-protection measures or changing covariates the focus has primarily been on comparing binary change (i.e. loss / gain) in forest cover or percentage deforestation at a pixel level (Andam et al. 2008; Geldmann et al. 2013; Butsic et al. 2016). This is then regressed with covariates at one point in time. The most commonly used covariates include land use, population density, road networks, rainfall, distance to towns, slope, altitude and agriculture potential. Alternatively, matching techniques are used to match pixels inside to outside protected area networks based on the covariates. These are used to compare binary change in forest cover at similar samples of pixels inside and outside the protected area i.e. control for differences in covariate information between inside and outside the protected area (Miteva et al. 2012).

When multiple land use/cover types (Beresford et al. 2013) or non-forest cover types (Carranza et al. 2014) are included in protected area evaluations, the lack of longer-term pre-protection measures for both the protected area and control area is a problem. For example, Naughton-Treves et al. (2005) reviewed 49 protected area effectiveness studies and found only two that included change over a time-frame greater than 20 years for inside and outside the protected area, with the average period being 13 years. This is because the detection of land use/cover change in habitats such as shrublands, grasslands, savannas and, especially, in mountain areas from satellite imagery remains a challenge (Pettorelli et al. 2016). Therefore, these habitats are often excluded as most studies are constrained by the goals of their experimental or quasi-experimental research design (Miteva et al. 2012). As a result, few studies incorporate a range of approaches to measure change including in situ measurement, surveys and interviews or monitoring data. Therefore, multiple land use/cover classes are rarely included and only a few studies include aspects related to water or to cultural ecosystem services (Naughton-Treves et al. 2005; Geldmann et al. 2013).

1.3 Why socio-ecological context is important

There is merit in “big remote sensing data” studies for determining protected area impact primarily because large sample sizes and spatial coverage can be achieved. Studies which rely on remotely-sensed data also allow for rigorous statistical modelling and matching methods which aid comparisons between different areas. These studies go a long way to providing a general impression of the effectiveness of protected areas. However, such generalizations may not allow for a full understanding of local-level impacts or may be impractical for informing local-level conservation decisions. This is because of the diversity of social and ecological contexts within which protected areas exist. Generalities determined at aggregated levels of analyses may also not pertain to individual protected areas. Alternatively, the effects of individual protected areas may be diluted at aggregated levels of analyses (Pressey et al. 2015; Ament & Cumming 2016; Eastwood et al. 2016).

Authors are beginning to acknowledge the importance of high-resolution and fine-scale detailed information when assessing spatially-explicit phenomena such as the influence of protected areas on land use/cover change for informing management and decision making at different scales. For example, Ament & Cumming (2016) showed a misalignment of patterns of land cover change at a national level with those at the local level. They suggested that, rather than focusing on discovering general rules about national or regional trends in and around protected areas, future research should start with finer-scale analyses at specific localities (Ament & Cumming 2016). Miteva et al. (2012) highlighted that regardless of whether rigorous experimental or quasi-experimental designs are used, baseline data on socio-economic and environmental factors are necessary for evaluations of protected areas. The importance of the local social, biophysical, economic and the temporal context of individual protected areas needs to be accounted for within protected area effectiveness assessments. This is because a number of factors influence the establishment of a protected area and confound protected area impact (Eastwood et al. 2016). In order to ensure that evaluations use the "right" data at the appropriate scale of analysis, interdisciplinarity, especially incorporating social and natural science approaches, is required (Miteva et al. 2012; Bennett et al. 2017).

Social science studies that focus particularly on conservation or environmental management are valuable for descriptive, diagnostic, disruptive, reflexive, generative, innovative, or instrumental reasons. However, the integration of social science insights into conservation practice is still limited (Bennett et al. 2017). This is particularly the case in the protected area effectiveness and impact evaluation literature. For example, Geldmann et al. (2013) reviewed 2599 publications and found 76 studies that used a counterfactual approach. Only five of these studies used in situ data collection including interviews and questionnaires or ecological, plot-based methods. The majority (89%) of studies used satellite remote sensing techniques with a focus on changes in forest cover. The high reliance on large-scale remote sensing or other course scale datasets to generate descriptions of counterfactual conditions, in contrast to landowner surveys or in-depth interviews accompanied by mixed methods analyses, is a noticeable indicator of the limited engagement with the social sciences.

The concept of counterfactuals and counterfactual thinking offers an important gateway for integrating social science methodologies into protected area impact evaluations. This includes the use of mixed qualitative and quantitative methods. However, tensions typically exist between disciplinary boundaries in terms of what type of approach, and therefore information and analyses, can generate credible claims about counterfactual conditions. For example, estimating counterfactual conditions for assessing protected area impact from non-experimental data is viewed as challenging and fraught with pitfalls. In addition, establishing a credible claim of causality using only qualitative data is viewed as more difficult than is the case with quantitative data (Ferraro & Hanauer 2014a). Therefore, understanding counterfactual conditions is dependent on the researcher and the limits imposed within their own disciplinary boundaries.

1.4 Boundary frameworks and concepts for achieving trans- and interdisciplinarity

Interdisciplinarity is about creating novel insights by thinking across disciplinary boundaries. Formally, it involves the combining of two or more academic disciplines into a research activity, solving a problem or addressing a topic that is too broad or complex to be dealt with adequately by a single discipline (Klein & Newell 1997). Transdisciplinarity includes interdisciplinarity but also includes engagement with actors outside of academia (Walter et al. 2007). Both of these research modes are also collectively known as integrated research

(Stock & Burton 2011). Three decades ago, conservation biology was defined as "an interdisciplinary science drawing on both the natural and social sciences and mixing art and science to make decisions and implement action" (Soulé 1985). Despite the early recognition in conservation biology, trans- and interdisciplinarity and, in particular, the integration of social science aspects has not fully been realised (Pooley et al. 2014; Bennett et al. 2017).

In contrast with conservation biology, which is relatively well established within academic institutes, sustainability science is a recent trans- and interdisciplinary academic field that aims to respond to threats within the Anthropocene. Sustainability science has its roots in the concept of "trans- and interdisciplinary research for sustainable development" which was introduced as a new form of research in the 1990s to provide sustainable solutions to complex problems (Klein 2004). Sustainability science as an overarching framework has been growing over the last two decades (Kates et al. 2001; Bettencourt & Kaur 2011; Kates 2011). It is now recognised by many as an evolving discipline in its own right (Jahn et al. 2012; Shahadu 2016) but also as a highly "undisciplinary" journey by early career researchers (Haider et al. 2017).

What distinguishes sustainability science from other fields is an attempt to generate knowledge and expertise on sustainability challenges while decreasing the distances between disciplines, theory and practice (Isgren et al. 2017). The study of socio-ecological systems within sustainability science has led to a wide range of frameworks that aim to structure research and explain or predict change. For a review of different frameworks see Binder et al. (2013), Scholz (2013) and (Cox et al. 2016). Despite multiple attempts to converge ideas in sustainability science, such as these latter review papers and associated initiatives and platforms, the research in the field is highly fragmented with different definitions, key concepts and theories informing, and being informed, by different research agendas (Brown 2016; Shahadu 2016). Therefore when embarking on a socio-ecological study, researchers are faced with numerous choices in terms of frameworks, concepts, and theory, which influence the design of a project and the analysis of the data (Rissman & Gillon 2017).

Concepts in sustainability science are strongly related but lack clarity on how they relate to each other. The degree of relational characteristics between concepts also differs among researchers (Shahadu 2016). Furthermore, rigidities and tensions exist between certain sustainability science domains or research groups. For example, some social scientists have suggested that scholars working on resilience thinking and theory attempt to solve problems such as poverty while drawing mostly on ecological concepts and theories and in so doing are insensitive to existing core social science concepts. The aim of providing a unifying theory for socio-ecological change is also seen as problematic when addressing complex phenomena associated with sustainability (Béné et al. 2011, 2014; Olsson et al. 2015). Resilience has evolved into an intricate and elaborate concept since the initial definition emerged in the ecological literature. The concept of resilience was initially derived from system theory and was defined as the ability of a system to bounce back or return to equilibrium following disturbance (Holling 1973; Béné et al. 2014) (Figure 1.2). It has undergone several iterations and is still being constantly refined (Béné et al. 2014; Biggs et al. 2015; Brown 2016).

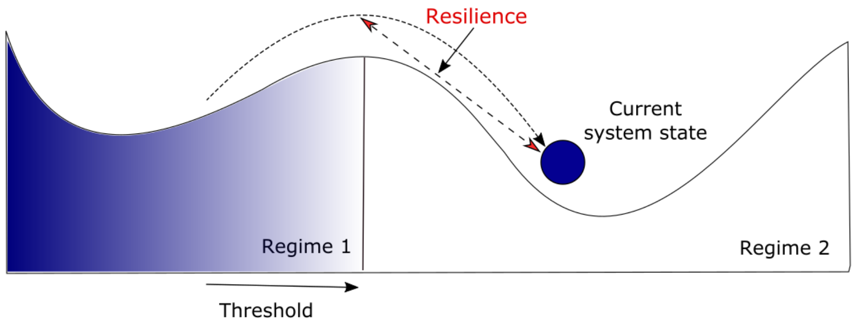


Figure 1.2 Resilience as a system property: the magnitude of change that a system can absorb without undergoing a regime shift i.e. remain in a particular stability domain (Holling 1973; Biggs et al. 2015).

For example, Folke (2016) drawing on the original ideas of Walker et al. (2004), defines resilience as: "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, and feedbacks, and therefore identity, that is, the capacity to change in order to sustain identity; resilience is a dynamic concept focusing on how to persist with change, how to evolve with change". Therefore, resilience is described as persisting with change in the "same basin of attraction" but adapting, improving and innovating within that basin. However, in cases when the resilience of a system becomes too robust or rigid there is a need to break resilience to enable shifts to new systems i.e. "completely new basins of attraction" (Folke et al. 2010; Folke 2016). The basis of resilience is the notion of self-organisation whereby systems occur in self-organising regimes with limits on how much the system can change before becoming a different type of system (Biggs et al. 2015).

In contrast to resilience thinking and theory, the Social-Ecological Systems Framework (Ostrom 2009) provides a framework for organising variables involved in different theories and models of social-ecological systems (Figure 1.3). It provides a common set of variables for studying a single social-ecological system or for comparing similar social-ecological systems (McGinnis & Ostrom 2014). It is theory-neutral to the extent that it can be applied using a variety of approaches and theories and hence supports the development of different types of models (Schlüter et al. 2014). It has been mainly used for discrete resource units e.g. fish drawn from a fishery, however, since its revisions and updates it is increasingly being used for a variety of ecosystem services and social-ecological systems (McGinnis & Ostrom 2014). The generality of the Social-Ecological Systems Framework and the fact that processes are represented as relation types allows one to link variables in any way reflecting different theories or empirically based assumptions of the dynamics of the social-ecological system. This allows for a combination and comparison of different approaches and theories to study a given social-ecological system and therefore one way to achieve a more multifaceted insight into their dynamics (Schlüter et al. 2014).

Despite the generality inherent in the Social-Ecological Systems Framework, it is important to acknowledge that socio-ecological research emerged from both natural and social science, and as a result, existing frameworks are often biased in promoting certain types of methods or variables for analysis. Therefore the choice of social and ecological variables and methods for linking them in the existing socio-ecological literature reflects diverse disciplines, epistemologies, and applications (Binder et al. 2013; Rissman & Gillon 2017). These challenges are common to diverse socio-ecological frameworks including resilience approaches and the Social-Ecological Systems Framework described above. Resilience thinking and theory originated from natural scientific inquiry, and as a result, it is largely biased towards natural science theory and methods (Folke et al. 2010; Biggs et al. 2015; Folke 2016). In contrast, the Social-Ecological Systems framework builds on the foundation of work conducted by social scientists. It is largely influenced by institutional analysis and development framework and has been used for microanalysis of a diverse range of social dilemmas mostly concentrated on common property natural resource management (Anderies et al. 2004; Ostrom 2009; Cox et al. 2010; Binder et al. 2013). Each of these different approaches suggests different variables and methods for analyses, which in turn influences findings and applications (Rissman & Gillon 2017).

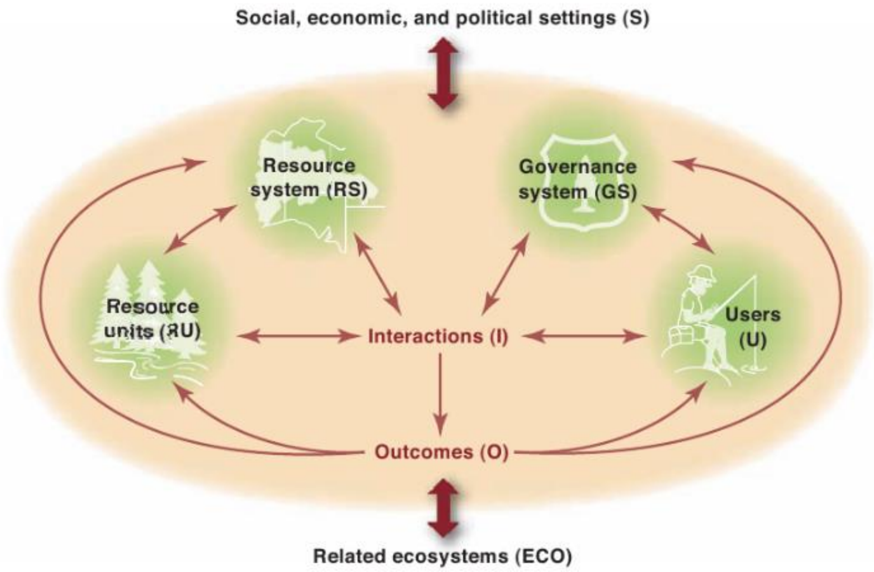


Figure 1.3 The core subsystems in the Social-Ecological Systems Framework proposed by Ostrom (2009)

In contrast to resilience approaches and the Social-Ecological Systems Framework, certain concepts related to the fields of sustainability science, conservation biology and land change science transcend boundaries and can be easily integrated and used across disciplines and across different frameworks. Examples of relevance to conservation biology include the concepts of ecosystem services, socio-ecological systems and land use transitions.

Ecosystem services include a wide variety of benefits that people derive from functioning ecosystems (Costanza et al. 2014, 2017). These include i) provisioning services, such as water, food, timber and fibre; ii) regulating services, that regulate ecosystem processes such as the climate and the water cycle e.g. water flow regulation; iii) cultural services that provide recreation, aesthetic or spiritual benefits; and iv) supporting services, which include processes such as soil formation, photosynthesis and nutrient cycling (MEA 2005). The concept was popularised by the MEA (2005) in 2005 (Figure 1.4) and has since been applied in countless studies as a concept, approach and framework for understanding the benefits that humans obtain from ecosystems (Costanza et al. 2017).

Initially the ecosystem services approach was largely associated with economic valuation and market related tactics. The aim of this approach was to generate payments for ecosystem services to re-invest back into the natural environment. However, the pre-occupation with placing a monetary value on ecosystem services has dissipated to some extent due to the limitations associated with monetarising certain ecosystem services. Complications arising from payments for ecosystem services programmes also exist. The ecosystem services concept still includes various forms of economic valuations, but it has been expanded to connect ecosystem services to human well-being without economic valuation (Costanza et al. 2017). Intrinsic values of nature have also now been incorporated to a certain degree (Pascual et al. 2017). The concept of ecosystem services is well established in the field of conservation biology as well as sustainability science and land change science. It has further been linked to climate change adaptation in the form of ecosystem-based adaptation (Brown 2016).

Figure 1.4 Original ecosystem service diagram presented in (MEA 2005). Since this, the diagram has been modified and used in countless publications and incorporated within numerous frameworks and related concepts and terminologies.

Socio-ecological systems are interdependent and linked systems of people and nature, which are nested across scales (Folke & Berkes 1998; Walker & Salt 2012) (Figure 1.5). Similarly, to the concept of ecosystem services, the term socio-ecological systems has been used in numerous publications as a lens and approach for analysing the relationship between humans and the environment as a coupled system (Brown 2016). As a lens, it focuses on humanity's dependence on nature but also the growing influence upon it. The concept of socio-ecological systems has proven especially pivotal within sustainability science and has provided a powerful analytical frame for understanding the inter-linked dynamics of environmental and societal change in countless case studies. The link to the ecosystem service concept and framing is obvious. However, in contrast to the concept of ecosystem services, the concept of socio-ecological systems is not as well defined in the sense that it is less rigid (Fischer et al. 2015).

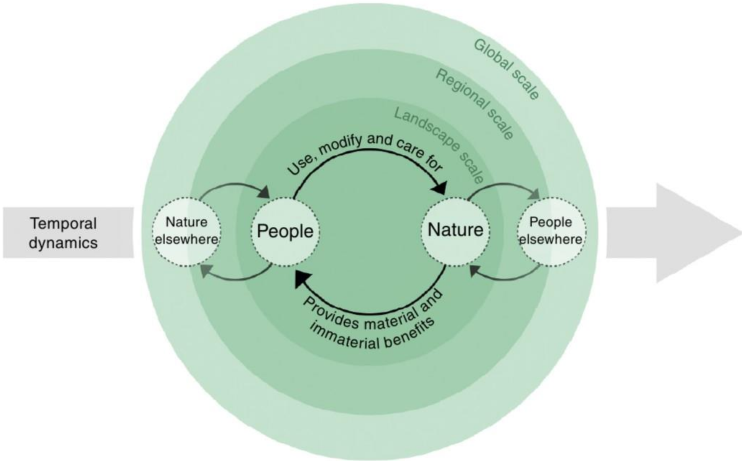


Figure 1.5 A diagram representing socio-ecological systems which are interdependent and linked systems of people and nature nested across scales (Fischer et al. 2015).

Land use transitions refer to any change in land use and thus cover from one state to another (Figure 1.6). Transitions in land use are viewed as reflecting multiple and reversible dynamics (Foley et al. 2005). One example could be when a system dominated by annual crops for local consumption is transformed to a large tree plantation in response to market demand. The concept can be used at multiple scales. This includes both at large scales, with a focus on slow and gradual processes of change, and at finer scales, where the focus is on local communities or agents at the individual level. In the case of the latter, abrupt transitions can often result from the adoption of new land use practices in response to certain critical events (Lambin & Meyfroidt 2010). Depending on the scale and area, different transition stages will be found which are dependent on history, social and economic conditions, and ecological context (Foley et al. 2005). Therefore, the concept highlights that land use change is non-linear and associated with other societal and biophysical system changes. Several typical land use transition pathways have been described. These have been linked to several different explanatory frameworks which draw on numerous case studies (Lambin et al. 2001; DeFries et al. 2004; Lambin & Meyfroidt 2010, 2011; Meyfroidt et al. 2013).

The ecosystem services, socio-ecological systems and land use transition concepts provide ideal framings for understanding protected area impact within a socio-ecological context. The integration of both ecosystem services and socio-ecological systems is inherent within the concept of land use transitions (Foley et al. 2005). For example, socio-ecological systems are associated with bundles of ecosystem services that interact and are influenced by scale-dependent as well as cross-scale effects and feedbacks (Biggs et al. 2015; Brown 2016). Land use is one way landowners interact with the environment to utilise ecosystem services (DeFries et al. 2004; Foley et al. 2005; Costanza et al. 2014). Different land use types are embedded within specific socio-ecological configurations and land use change occurs through a series of societal and biophysical changes driven by multi and cross-scale socio-economic dynamics and socio-ecological feedbacks (Lambin & Meyfroidt 2010). Although, land use practices are essential to humanity because they provide critical natural resources and harness ecosystem services, some land use practices may also be detrimental to the environment. This is because depending on the type of land use and ecosystem, land use practices have the potential to degrade a subset of critical ecosystem services while prioritising others (DeFries et al. 2004; Foley et al. 2005). Protected areas have been one tool used by conservationists to either remove or reduce these threats (Cumming et al. 2015; Cumming 2016).

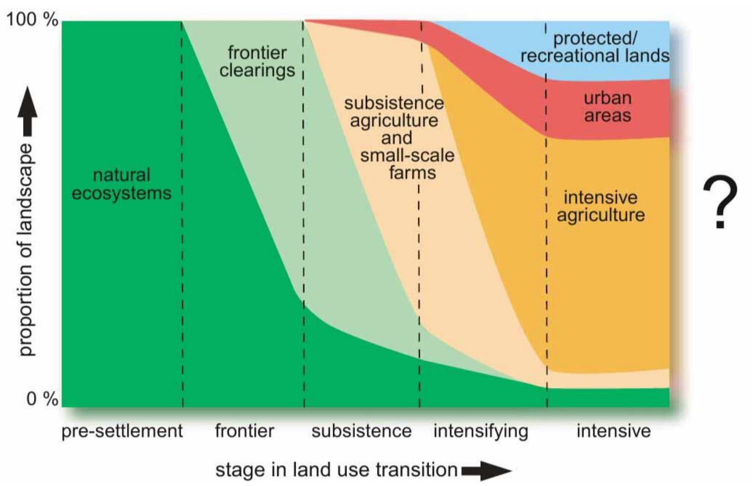


Figure 1.6 General land use transitions that may be experienced in an area over time. Different areas will be in different transition stages and not all areas will move linearly through these transitions. Some places remain in one stage for a long period of time while others move rapidly through stages (Foley et al. 2005).

The concepts of socio-ecological systems, ecosystem services and land use transitions are interlinked and provide useful framings for understanding change within a systems perspective. Ecosystem service trade-offs are inherent in land use transitions (Foley et al. 2005). Accordingly, the causal mechanisms which drive trade-offs require careful consideration for sustainable land use policies. Change can be exogenous to the socio-ecological system, and driven by socio-economic forces such as globalisation, and/or endogenous and driven by socio-ecological feedbacks associated with past land use, land degradation or inherent ecological constraints (DeFries et al. 2004; Lambin & Meyfroidt 2010). Economic opportunities are considered the main driver of land use change and there is limited evidence that societies retreat from natural ecosystems because they have been degraded. This is because economic opportunities allow for institutional or technological innovations that cause shifts in land use that accommodate environmental constraints (Lambin et al. 2001; Lambin & Meyfroidt 2010). Determining the role played by socio-ecological feedbacks versus broader socio-economic dynamics is important for understanding and modelling socio-ecological change with implications for sustainable land use and conservation (Lambin & Meyfroidt 2010).

Despite the linkages between the concepts of ecosystem services, socio-ecological systems and land use transitions described above, there is a need to situate these concepts within an overarching approach that advocates the use of different concepts to understand socio-ecological change and that also integrates the importance of multiple methods, knowledge streams and openness in research approaches. Accepting that multiple methods, knowledge streams and theories can be relevant within a single research project is typical of trans- and interdisciplinary studies when characterised by pluralism, a boundary concept recommended by social scientists (Olsson et al. 2015). Pluralism is an approach for achieving integrated research and therefore involves combining disciplines to contribute towards understanding complex phenomena or to solve sustainability challenges (Isgren et al. 2017). However, pluralism attempts to avoid insensitivity to methodological and theoretical developments in any field. This is because the goal of a pluralistic approach to research is to rely on multiple theories, methods and inference approaches as required. Such approaches can be drawn from many disciplinary fields as opposed to using one theory or method to cross boundaries (Olsson et al. 2015). Sustainability science scholars advocate methodological pluralism for research but also call for procedural rigor and the necessity for continuous research (von Wehrden et al. 2017).

1.5 The impact of protecting mountain ecosystems

Mountains are recognised as global and regional water source areas which are critical for providing flows of water for biodiversity, ecosystem function and downstream communities and economies (Viviroli et al. 2003, 2007, 2011). The ability of mountain headwaters to capture, store and release water in a manner that it is maintained during a range of flow conditions, especially during mid to low flow periods and during dry years, is considered as one of the most vital services of these ecosystems. Progressive loss of streamflow results in serious risks to human well-being (Postel & Thompson 2005; Xiao et al. 2015; Viglizzo et al. 2016). Well-managed mountain catchments contribute significantly to maintaining streamflow. This includes their ability to regulate extreme high flows and to ensure streamflow under all flow conditions, especially during low flow conditions and dry years when demand for water increases (Garrido & Dinar 2009; Xiao et al. 2015; MacKinnon 2016).

1.5.1 Land use/cover change in mountain ecosystems

Mountainous regions have globally experienced large-scale farmland abandonment over the last 50 years (MacDonald et al. 2000; Lasanta et al. 2006b; Nyssen et al. 2009; Queiroz et al. 2014). This has included a decline in traditional labour intensive and subsistence agricultural practices and, in general, marginal agricultural land has been abandoned (MacDonald et al. 2000). Causal mechanisms of farmland abandonment have been context specific, interrelated and influenced at many scales. Farmland abandonment during the second half of the 20th century, particularly in mountainous areas, has been linked to the collapse of certain rural societies, migration to urbanized areas, low land productivity, political changes and incentive measures to cease farming (MacDonald et al. 2000; Pereira et al. 2005; Keenleyside et al. 2010; Beilin et al. 2014).

Land abandonment has significant environmental, landscape and socio-economic implications and can lead to an increase in vegetation cover of certain cover classes (Lasanta et al. 2015). Abandoned lands occupy a large area and therefore management of abandoned lands has important consequences for biodiversity and ecosystem services. However, farmland abandonment is a conservation dilemma with society and scientists debating the resultant impacts on biodiversity and ecosystem services. There is also no agreement on suitable management approaches which should be used. For some, land abandonment is an opportunity for passive restoration to improve the habitat for species that were affected by agricultural use. Others propose active intervention in the landscape in order to control the negative effects of re-vegetation on biodiversity and ecosystem functioning related to water supply and fire risk among other related services (Queiroz et al. 2014; Lasanta et al. 2015).

Farmland abandonment is expected to intensify over the next century and spread to other parts of the world (Figueiredo & Pereira 2011; Plieninger et al. 2014; Queiroz et al. 2014; Lasanta et al. 2015; Price et al. 2015). Despite observed trends, existing farmland abandonment studies and models do not consider social and cultural factors that may encourage the continuation of uneconomic or semi-economic recreational and tourism activities in mountain lands and the influence of these activities on views and values of mountain conservation. In particular, the consideration of how drivers of land use change interact with cultural ecosystem services, such as personal nature-based recreation and ecotourism, and how this is associated with the process of farmland abandonment in mountain regions is limited (Keenleyside et al. 2010; Figueiredo & Pereira 2011; Queiroz et al. 2014; Price et al. 2015; Hinojosa et al. 2016; Holman et al. 2017). Certain areas that are listed as abandoned may in fact be areas that are subject to very low levels of management or be under management not necessarily evident in methods used to determine land use trends, such as remote sensing (Keenleyside et al. 2010; Beilin et al. 2014). Given trends and projections in farmland abandonment in mountain regions and the challenge this poses for conservation, there is a need to improve our understanding of socio-ecological system configurations in mountain systems, especially in understudied regions (MacDonald et al. 2000; Lambin & Meyfroidt 2010).

1.5.2 The impact and value of conservation initiatives in mountain environments

Numerous conservation initiatives and interventions have targeted mountain areas. These include the establishment of formal protected areas but also payment for ecosystem services programmes. Many of these interventions including protected areas in mountain ecosystems have been established to protect the sustainable flow of water for ecological and downstream human benefits (Gaveau et al. 2009). In addition, most were established as wilderness areas for cultural-based ecosystem services associated with recreation or inspired by the intrinsic value of nature and wild natural landscapes (Cumming et al. 2015).

Establishing a relationship between conservation initiatives in mountain areas and changes in land use/cover and ecosystem service provisioning has been challenging for research. For example, evidence of the delivery of water-related ecosystem services from payment for ecosystem services schemes has proved elusive because of difficulties in measuring and attributing changes in the provision of water-related ecosystem services to specific interventions (Landell-Mills et al. 2002). Porras et al. (2008) identified and reviewed 95 payment for ecosystem services schemes for watershed management in developing countries and found that integrated approaches to understanding change were limited. Specifically, on-site measurements and modelling of land use and water relationships was severely limited and in most schemes impacts were only based on views of users, local people and adminstrators. The difficulty in measuring change in mountain ecosystem services is also reflected in that only 0.04% of the 293 papers on services from ecosystems published in *Ecosystem Services,* since its launch, have been on mountain ecosystems (Costanza et al. 2017).

In addition to a limited focus on mountains, studies on ecosystem service dynamics seldom incorporate historical analyses and an understanding of temporal dynamics has largely been absent (but see Dallimer et al. (2015), Renard et al. (2015), Sutherland et al. (2016)). Instead, studies have focused on exploring the spatial patterns of ecosystem service interactions and spatial trade-offs in protected areas and non-protected sites as opposed to temporal changes (Chan et al. 2006; Naidoo & Ricketts 2006; Maes et al. 2012; Onaindia et al. 2013; Garcia-Llorente et al. 2015; Eastwood et al. 2016; Xu et al. 2017). Alternatively, existing assessments have focused on measuring the effects of management activities on water-related ecosystem services such as alien plant clearing (Le Maitre et al. 2014), afforestation or deforestation (see Bosch & Hewlett (1982), Brown et al. (2005), and Moreno & Oechel (2012) for reviews) and fire (Lindley et al. 1988; Scott & Van Wyk 1990, 1992; Scott 1997). There has also been a focus on measuring the effects of planned interventions or anticipated land use/cover change on ecosystem services (Hodder et al. 2014; Quintas-Soriano et al. 2016). For example, many studies have investigated the impacts of land use/cover change on hydrology, often using scenario approaches or comparing relatively pristine before and after cases or side-by-side cases (Li et al. 2009; Savary et al. 2009; Warburton et al. 2012; Sajikumar & Remya 2015; Weyer et al. 2015; Xiao et al. 2015; Prucha et al. 2016; Schütte & Schulze 2017; Wang et al. 2017).

When studies have included temporal change in ecosystem services, pre-protection measures have not been included in the assessments (Brill et al. 2017). A non-mountain case study exception is that of Zorrilla-Miras et al. (2014) which investigated change in land use and cover inside and outside a coastal marsh protected area in Spain from 1918-2006. In this study, existing literature and perceptions of beneficiaries were used to determine the effects of land use and cover change on ecosystem service delivery. The results showed an increase in high economic value irrigated agriculture or aquaculture at the expense of regulating services such as hydrological regulation outside the protected area, while regulating and cultural services were described as being delivered mainly inside the protected area (Zorrilla-Miras et al. 2014).

When studying the value placed on protected areas, many studies have focused on the importance of direct economic use values and especially recreational and spiritual or cultural values (Chan et al. 2006; Daniel et al. 2012; Maciejewski et al. 2014; Ament et al. 2016). Option, existence and bequest values express people's appreciation of the continued existence of species and ecosystems beyond their immediate use. These are referred to as non-use values in the ecosystem services literature. Intrinsic values involve the ethical considerations humans make about other people's and other species' rights to live (Harris & Roach 2015; Costanza et al. 2017). Few studies appreciate the potential of intrinsic-moral and non-use economic values for building societal support for protected areas (Chan et al. 2012b). Although the focus on non-use economic values has been increasing (Costanza et al. 2017), this has not been the case for intrinsic values. For example, Vucetich et al. (2015) reviewed the conservation related literature over the last two decades and found only 18 papers that focused on intrinsic value. These papers were on average only cited once a year from 2009-2013. This is despite intrinsic value being fundamentally important to conservation (Vucetich et al. 2015).

1.5.3 Determining the impact of protection on water-related ecosystem services

Interpreting the impact of protected areas on streamflow requires an understanding of interactions among multiple variables (Van Dijk et al. 2012; Xiao et al. 2015; Schäfer et al. 2016). This includes the way in which water moves from rainfall to above- and below-ground water flows and how long-term changes in land use and cover (land use/cover) interfere with water movement through a catchment (Brown et al. 2005; Brauman et al. 2007; Wang-Erlandsson et al. 2014). The influence of land use change on water flows is complex and context specific. At times, transformed areas can still operate in a manner comparable to natural conditions and thereby maintain streamflow as required for human needs. This depends largely on management and land use practices and the influence of land use and cover changes on water budget partitioning in terms of soil evaporation, transpiration, baseflow and overland flow (Nunes et al. 2011; Van Dijk et al. 2012; Schäfer et al. 2016). However, the important factors influencing water flow and how these factors interact over time differs significantly in arid and humid areas and is highly linked to precipitation and above-ground biomass (Nosetto et al. 2012; Viglizzo et al. 2016).

Streamflow is made up of all the water generated from a catchment and consists of stormflow and baseflow. Stormflow occurs at or near the surface of a catchment and is generated from a specific rainfall event. Baseflow includes the contribution to streamflow from previous rainfall events where rainfall has percolated through the soil horizons and contributes as delayed flow to streams. Baseflows are considered "dry weather" flows, which are significant in maintaining flows under low flow conditions (Schulze 2008a). Changes in above-ground vegetation biomass are linked to a range of land cover changes and management practices which result in variable impacts on water flows depending on the local context (Lindley et al. 1988; Brown et al. 2005; Feikema et al. 2013; Schäfer et al. 2016; Viglizzo et al. 2016).

The reason that above-ground biomass has been highlighted as important in terms of regulating streamflow is that it is directly linked to water infiltration through the soil surface and subsequent percolation of water through the soil. These two processes are important in determining how much rainfall flows off immediately over the surface versus how much is retained in the soil and released more slowly over a lagged time (Brauman et al. 2007; Le Maitre et al. 2014). Above-ground biomass is also related to the amount of water lost through interception, transpiration and soil evaporation (Lin 2010; Moreno & Oechel 2012; Van Dijk et al. 2012; Feikema et al. 2013). The benefits of increased above-ground biomass on water flows have been well described in the literature. However, the benefits can vary depending on climate, the type of vegetation as well as the particular land use and management practices in the area (Brown et al. 2005; Moreno & Oechel 2012; Heath et al. 2014; Schäfer et al. 2016).

South Africa in particular has a significant literature on the hydrological role of vegetation in mountain catchments (Wicht 1949; Kruger & Wicht 1976; Le Maitre et al. 2002; van Wilgen 2009; Dye 2013; Witt 2014; Bennett & Kruger 2015). This research draws largely on a comprehensive set of paired catchment studies that have been used to assess the impacts of, for example, afforestation, harvesting and fire on streamflow (Wicht 1971; Van Wilgen 1981; Bosch & Hewlett 1982; van Wilgen et al. 1992; Scott 1993, 1997; Dye 1996; Le Maitre et al. 2014). In general, findings from paired catchment studies point towards an increase in streamflow in proportion with the magnitude of the disturbance primarily measured in terms of the amount of reduction in forest and in general vegetation cover (Van Wilgen & Kruger 1985; Bosch et al. 1986; van Wilgen et al. 1992; Brown et al. 2005; Moreno & Oechel 2012).

The increase in streamflow is viewed as being due to decreased transpiration losses and increased overland flow during large rainfall events because of the loss of vegetation, surface cover and post-fire soil hydrophobicity. Decreased transpiration losses, in turn, are due to the loss of vegetation in the post-disturbance environment (Lindley et al. 1988; Scott & Van Wyk 1990; Scott 1997; Meixner & Wohlgemuth 2003). Increases in streamflow are mostly comprised of stormflow, which is overland or shallow flow that leaves the mountain during or shortly after rainfall events (Bladon et al. 2014). Empirical small paired-catchment studies in fynbos have shown increases in streamflow after fire to be short-lived, ~1–2 years (Lindley et al. 1988; Scott 1993). However, the extent and intensity of disturbance across multiple catchments can influence the size and timing of increases in streamflow after disturbance events (Shakesby 2011). Furthermore, an increase in the intensity and occurrence of a disturbance could reduce the window for vegetation recovery and result in permanent change to hydrological control in mountain catchments (Bladon et al. 2014).

1.6 Pluralism in studying protected area impact, using a counterfactual framing

A multitude of variables can influence the impact of conservation in mountain environments and there are numerous challenges in measuring and attributing these impacts. The use of counterfactual framings is important, but the counterfactual conditions should be informed by numerous integrated approaches to understanding change (Naughton-Treves et al. 2005; Fox et al. 2006; Miller et al. 2008; Roy et al. 2013; Bennett et al. 2017).

Although existing socio-ecological system frameworks are generally compatible with multiple theories, these frameworks focus on diagnosing a system in terms of resilience, vulnerability or some other system property rather than informing an integrated process to assess the impacts of an intervention within a system. In terms of trans- and interdisciplinary processes, the role of integrated research and knowledge in existing socio-ecological system frameworks remains unclear i.e. how to incorporate multiple disciplines, methods and knowledge sources in a manageable way. Therefore, existing socio-ecological system frameworks do not provide guidance on where trans- and interdisciplinary approaches fit within the overall framework and how these can be explicitly used in a pragmatic way, particularly from an individual researcher perspective. There is also a lack of explicit identification of where and when stakeholders can be engaged. Although deceptively simple, few conservation researchers take the trouble to initiate dialogue with one or more conservation, management or non-governmental organisations engaged broadly in the area of work to determine what is most pressing and relevant (Laurance et al. 2012).

Existing socio-ecological frameworks also lack a dynamic temporal component. Many include "history" as a variable within the framework rather than as a dynamic process that has resulted in the current situation in the landscape. The history of the landscape is a key confounding factor under a counterfactual framing. An exception is the framework presented by Cumming et al. (2015) which integrates ideas from resilience with Ostrom (2009)'s Social-Ecological Systems framework and explicitly incorporates both temporal and spatial scales. The framework is well suited for determining the resilience of protected areas by integrating socio-ecological feedbacks and cross-scale effects that often dominate the dynamics of protected areas and other socio-ecological systems. However, the incorporation of processes inside and outside-protected areas within a counterfactual approach is not explicit and the scope for pluralistic methodologies is not clear. Therefore, existing social-ecological frameworks from sustainability science do not provide ready mechanisms for promoting integrated research when studying protected area impact. As a result, there has been limited integration between existing socio-ecological frameworks and counterfactual framing ideas from protected area impact evaluations.

This thesis uses pluralism as an integrative framework for promoting trans- and interdisciplinarity while using the concepts of socio-ecological systems, ecosystem services, land use transitions and counterfactuals to assess the impact of over ~40 years of mountain protection, drawing on comparisons of ~30 and ~40 years before and after protection respectively, with an adjacent area of similar terrain as the counterfactual. I build on the existing progress made by conservation scientists in the Cape Floristic Region in terms of integrating ecosystem services into conservation assessments and planning as well as using trans- and interdisciplinary approaches in conservation research (as described in Section 1.1). I address three research gaps that are hampering more effective integration between the protected area impact evaluation literature and sustainability science. These include i) the relationship between socio-ecological perspectives of protected areas and protected area impact evaluation; ii) the limited integration of social science mixed methods analyses and place-based level analyses in protected area impact evaluations; and iii) the lack of guidance on how to incorporate trans- and interdisciplinary approaches when applying a socio-ecological framing to understand the influence of protected areas using a counterfactual approach, particularly from an individual researcher perspective.

I draw on the concepts of socio-ecological systems, ecosystem services and land use transitions as opposed to existing socio-ecological frameworks such as the Social-Ecological Systems Framework and resilience approaches. This is because these three concepts are not restrictive in their approaches to assessment and therefore can be easily integrated into a counterfactual framing as informed by progress made in the protected areas impact evaluation literature. I specifically use the concepts of ecosystem services, land use transitions and socio-ecological systems due to their inherent connections as described earlier in Section 1.4 and their ability to transcend the fields of conservation biology, sustainability science and land change science. I draw on pluralism as an integrating mechanism between these concepts and as the foundation to incorporate multiple approaches into the research methods applied. Most of the literature on integrated research focuses on using teams to achieve trans- and interdisciplinarity. Because of this, achieving plurality within research from an individual researcher perspective has not been the focus (Pooley et al. 2014).

1.7 Thesis overview

This thesis focuses on understanding the effects of mountain protection on ecosystem services over time and how this influence relates to broader socio-economic and ecological drivers of landscape change.

1.7.1 Research aim

To use a counterfactual framing informed by pluralism in its methodology to determine the impact of protected area establishment in a mountain catchment important for regional water supplies in the Cape Floristic Region, South Africa.

1.7.2 Research objectives

1. Determine the effect of mountain protection on land use/cover, vegetation, fire and streamflow accounting for broader socio-economic and ecological drivers of landscape change.
2. Investigate what other types of values (economic, including use and non-use, and intrinsic) are important when determining the impact of mountain protected areas.

1.7.3 Thesis outline

Section 2.1 provides the socio-economic and ecological context to the broader study region and the case study area.

Section 2.2 describes the overarching thesis methodology with a focus on describing the approach applied to generate the counterfactual conditions used to validate protected area impact on vegetation, land use/cover, fire and water flows. I focus on how the counterfactuals were developed and used across the data sections of the thesis.

Section 3-5 operationalise the thesis methodology and address the thesis aim and objectives.

Section 3 contextualises socio-ecological change in the privately owned portion of the mountains and determines current landowner views and values of mountain protection and regulation. I focus on understanding the drivers of change in land use/cover over the last forty years, and more qualitatively over the last century. This section also assesses the relationship between the values that landowners placed on mountain protection and certain landowner and property characteristics.

Section 4 investigates the impact of mountain-protection on vegetation cover and composition over the last 40 years using before-after vegetation survey and ground photograph comparisons. I use the results of Section 3 as well as an analysis of climate and the results of an analysis of aerial photographs (in Section 5) to determine what would have happened if the protected area had not been established.

Section 5 assesses the influence of mountain protection on streamflow over the last ~50 years. I expand on the land use/cover change described in Section 3 by using orthorectified historical aerial photographs for 1949 and 1972 and current orthoimages for 2014. I develop land use/cover scenarios which represent alternative counterfactual conditions for the protected area. I then configure a hydrological model to determine the influence of land use/change patterns over the last 50 years on streamflow inside the protected area and how this differs to counterfactual scenarios of land use/cover. Data from Section 3 and 4 were used directly for parameterising the hydrological model.

Section 6 highlights the main arguments of the thesis, summarises and discusses the results of the thesis in relation to existing literature and then presents the broader significance of the research, outlines research needs and reflects on the success and challenges of the thesis methodology from an individual researcher perspective in the context of trans- and interdisciplinarity.

Section 2. Methods

2.1 Study area

2.1.1 Water source areas in South Africa and the Cape Floristic Region

South Africa is a water-limited country, prone to droughts. Water supply from rivers fluctuates widely because of the large inter- and intra-annual variation in the amount and distribution of rainfall (New 2002). Periodic droughts, associated with consecutive years of below average rainfall, have been seriously detrimental to the national economy over the last few decades (Rouault & Richard 2003; MacKellar et al. 2014). Water resources are becoming increasingly valuable due to increased demand by agricultural, industrial and urban sectors. Therefore, the country has already experienced increased levels of water scarcity, compounded by population growth and issues of social and economic development. Climate change will exacerbate these problems by exerting additional stresses on water resources (New 2002).

Water source areas situated largely in mountainous regions occupy eight percent of South Africa's land surface but supply 50% of mean annual runoff (Figure 2.1). These sources are critical for surface water supplies and for supporting local, regional and national economic activities. About 16% of water source areas in South Africa are under formal conservation protection, most of which is in the Cape Floristic Region in the south-western part of the country (Nel et al. 2013) (Figure 2.1).

2.1.2 The Cape Floristic Region

The Cape Floristic Region is a geographically confined region (90 760 km^2) in South Africa that includes considerable variation in climate and topographic conditions (Figure 2.1 and 2.2A). It is a global biodiversity hotspot and World Heritage Site but also a region where both urban population and irrigated agricultural land are increasing rapidly (Allsopp et al. 2014). Fynbos is the most distinctive and common vegetation type of the region. Fynbos is a fire-prone shrubland well represented, in terms of species diversity and biomass, by ericoids, proteoids and restioids. The area comprises ~9 000 species, 68% of which are endemic. The region, especially in the west, mostly experiences a Mediterranean-type climate with cool, wet winters and hot, dry summers. It is characterized largely by having nutrient-poor soils (Manning & Goldblatt 2012).

2.1.2.1 Existing mountain conservation policy in the Cape Floristic Region

Mountain catchments in the Cape Floristic Region are important water source areas for the urban, agricultural and industrial sectors (New 1999). As mentioned above, many of these mountainous areas are also under protection either formally via protected areas or informally under certain pieces of legislation or landowner agreements with the provincial conservation authorities, CapeNature.

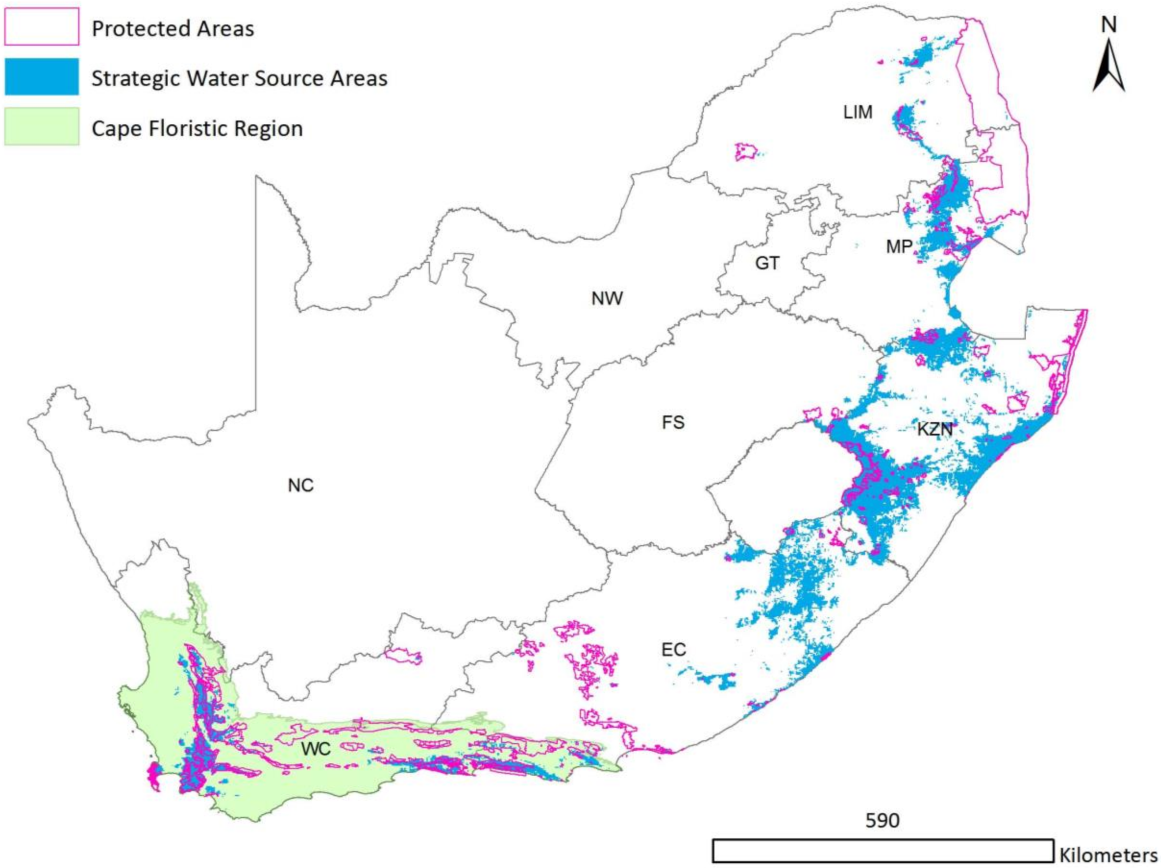


Figure 2.1 National strategic water source areas in South Africa (CSIR 2013; MDB 2013) and protected areas (DEA 2016) that contain portions of these areas. Most of the national strategic water sources areas in the Cape Floristic Region (SANBI 2012) are under protection in comparison to the rest of South Africa. Protected areas include Special Nature Reserves, National Parks, Nature Reserves, Protected Environments, World Heritage Sites, Forest Nature Reserves and Forest Wilderness Areas and Mountain Catchment Areas.

Thirteen protected area clusters (~10 947 km^2 in size) comprise the Cape Floristic Region World Heritage Site inscribed on the UNESCO World Heritage Site list (Figure 2.2B). These clusters represent outstanding universal value for biodiversity and uniqueness but also have high protection levels in terms of conservation (UNESCO 2015; van Wilgen et al. 2016a). Protected area clusters are largely situated in the upland catchments of the Cape Floristic Region. This is because of the historical legacy that afforded higher protection to mountainous land as opposed to lowland areas (Rouget et al. 2003a; van Wilgen et al. 2016a). Most of these catchments are also recognised as national strategic water source areas, playing a critical role in supporting a complex water supply system in the Western Cape of South Africa, comprising interlinked dams, canals, pipelines, tunnels and distribution networks (Nel et al. 2013a, 2013b). The official buffer zone to the Cape Floristic Region World Heritage Site is 7558 km^2 and largely includes privately owned mountainous land delineated under the Mountain Catchment Areas Act 63 of 1970 as well as other terrestrial and marine protected areas (UNESCO 2015) (Figure 2.2B).

Demarcated Mountain Catchment Areas (under the Mountain Catchment Areas Act 63 of 1970) are recognised as non-formal protected areas in terms of the National Environmental Management: Protected Areas Act 57 of 2003 and cover approximately 5400 km^2 of the Cape Floristic Region, making up 6% of the total protection (18%) (van Wilgen et al. 2016a). In 1998, the National Veld and Forest Fire Act 101 of 1998 replaced certain sections of the Mountain Catchment Areas Act to redirect the ownership and liability of fire management and control to private landowners in mountainous areas with assistance from community driven Fire Protection Associations (see more information below under 2.1.3.4). Land use, soil erosion and intruding plant directives, however, remain under the Mountain Catchment Areas Act (Strydom & King 2009; van Wilgen et al. 2016a). In general, there is no consensus of the national status of the Mountain Catchment Areas Act and there is no organisation actively administering or managing the Act (Rabie & Burgers 1997; Bennett & Kruger 2015; van Wilgen et al. 2016a).

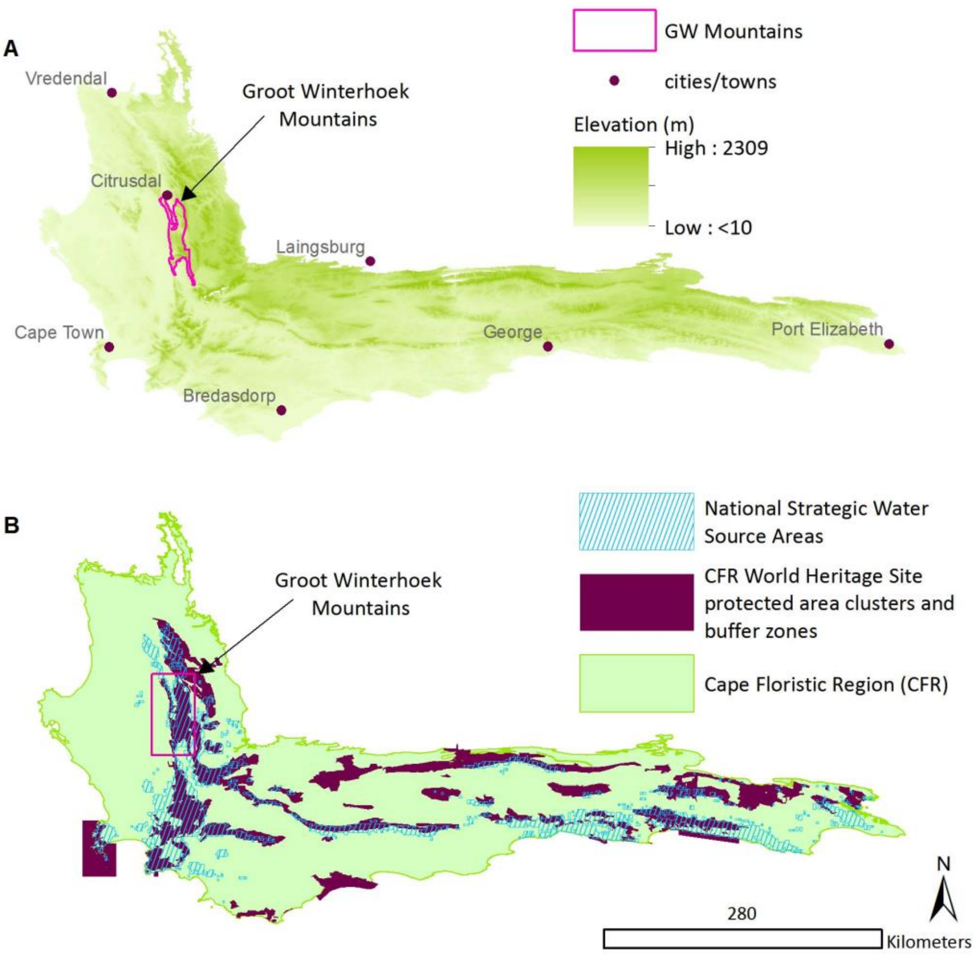


Figure 2.2 Study Area: The Groot Winterhoek Mountains with A) mean altitude: 1058 m and maximum elevation: 2077 m) and B) identified as a national strategic water source area in South Africa and comprising a protected area cluster and associated buffer area in the Cape Floristic Region World Heritage Site (CSIR 2013; UNESCO 2015; DEA 2016; NGI-DEM 2016). Please refer to Figure 2.1 for the wider geographical area within which the Cape Floristic Region is apart.

2.1.2.2 Historical context to conservation policy in the Cape Floristic Region mountains

Prior to the 1970s, there were serious concerns for mountain catchments in the Cape Floristic Region and securing proper management of mountains presented a significant challenge to scientists and government. The combination of burning and grazing as well as trampling were seen to have large-scale consequences for mountains and especially for the quality and quantity of the water from these catchments (Wicht 1943; Ross & Tempel 1961, 1961; Pooley 2012; Bennett & Kruger 2015; van Wilgen et al. 2016a).

To regulate and conserve mountain areas in the Cape Floristic Region, as well as more broadly at a national level, the government spent vast amount of time and money on policies and actions to protect and regulate private land use. This included expropriating land for the establishment of protected areas and developing a range of land use and management regulation policies for privately owned land. Initially, government's approach to the management of private catchments in mountainous wildlands was by appointment of Fire Protection Committees (in 1949) under the Soil Conservation Act of 1949. This then progressed through numerous unsuccessful Acts and related actions to the Mountain Catchment Areas Act 63 of 1970, which is still in place today (Strydom & King 2009; Bennett & Kruger 2015; van Wilgen et al. 2016a) and is recognised by South Africa's National Environmental Management Protected Areas Act of 2003.

2.1.3 The Groot Winterhoek Mountains

This thesis uses the Groot Winterhoek Mountains (~800 km^2; 32°38; 33°25´S and 18°56´; 19°16´E) in the south-western portion of the Cape Floristic Region to understand the long-term impact of mountain protection on land use/cover, vegetation, fire and streamflow over the last ~50 years (Figure 2.2 A). The Groot Winterhoek Mountains is a poorly studied region, mainly due to difficulty of access (Mucina & Rutherford 2011). There is no published literature on vegetation or land use/cover change for the catchment and there has been no focused research on these topics conducted in the area since the 1980s. The area has been identified as a national strategic water source area in South Africa and forms part of the Cape Floristic Region World Heritage Site (Figure 2.2 B).

The case study area specifically comprises the following areas (Figure 2.3) within the Groot Winterhoek Mountains:

- the **Groot Winterhoek Wilderness Area**, which includes i) land under strict formal protection (~200 km^2) which was privately owned prior to 1978; and ii) land that has been under strict protection since ~1910/15 (~100km^2); and
- **privately owned land** (~500 km^2) situated in the Groot Winterhoek Mountains adjacent and proximal to the Groot Winterhoek Wilderness Area. The area that is privately owned is surrounded by the boundary of the Groot Winterhoek Mountain Catchment Area as demarcated by the Mountain Catchment Areas Act of 1970.

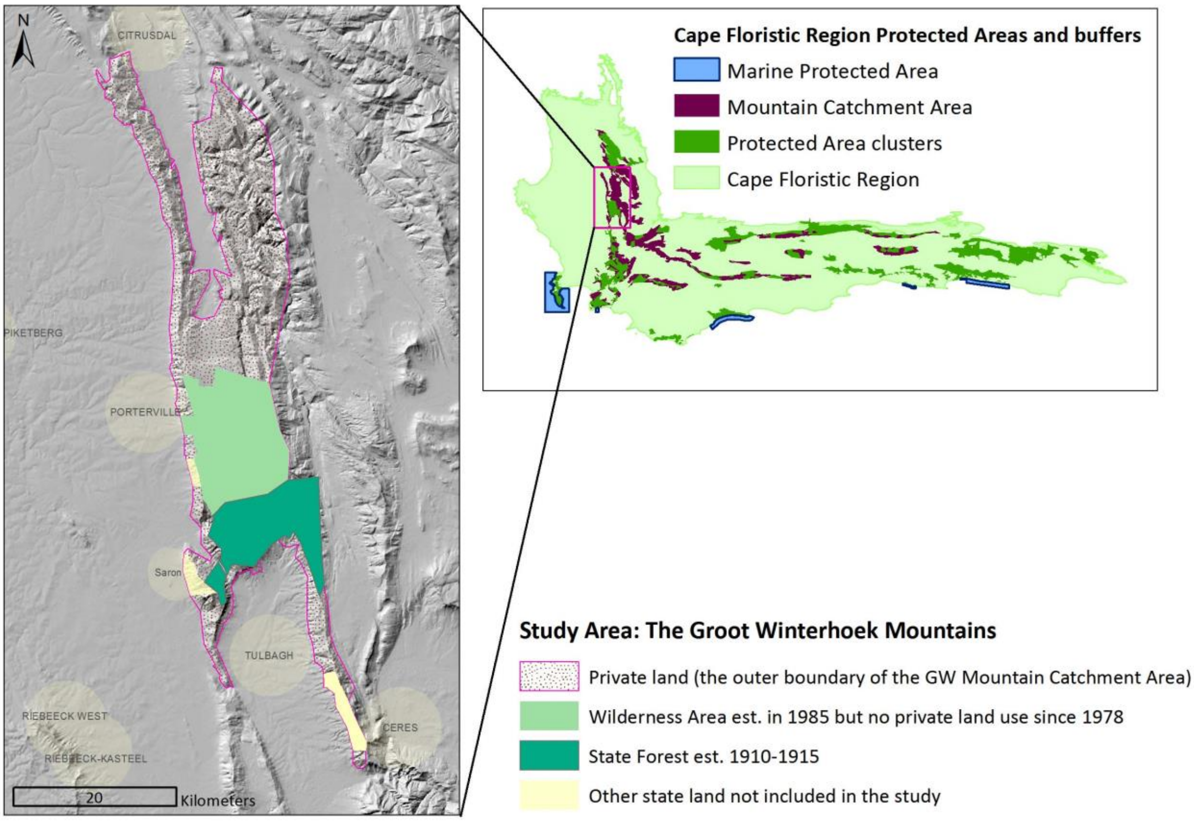


Figure 2.3 The study area: The Groot Winterhoek Mountains (SANBI 2012; SUDEM 2014; DEA 2016; SG 2016). Please refer to Figure 2.1 for the wider geographical area within which the Cape Floristic Region is apart.

2.1.3.1 Administrative and socio-political information

The Groot Winterhoek Wilderness Area (~300 km^2) was proclaimed in 1985 as a formal protected area under the Forest Act of 1968 (SA 1985) after a series of expropriations of privately owned land from the mid sections of the Groot Winterhoek Mountains between 1965 and 1978 (Bands 1985). Specifically, in the past, ~200 km^2 of the area was privately owned and used predominately for livestock grazing, wildflower and plant harvesting and small-scale cultivation (Bands 1985). The area was formally proclaimed a Wilderness Area (SA 1985) through a process of landowner negotiations and land expropriations which started in 1965. The last property was expropriated in 1978. The central aim of establishing the wilderness-protected area was to protect the water quantity and quality delivered from the catchment (Bands 1985). The Groot Winterhoek Mountain Catchment Area was demarcated in 1981 under the Mountain Catchment Areas Act 63 of 1970 in an attempt to reduce the amount of land that was required for further expropriation for formal protection (SA 1981).

CapeNature is the management agency for the Groot Winterhoek Wilderness Area. CapeNature is a public institution with the statutory responsibility for biodiversity conservation in the Western Cape. It is governed by the Western Cape Nature Conservation Board Act 15 of 1998. The overall purpose of the wilderness-protected area is "*To conserve the cultural and natural heritage of the Groot Winterhoek Wilderness and supply quality water, sustainable access, tourism and other socio-economic benefits through landscape level* *partnerships"* (CapeNature 2016a). CapeNature is also mandated to administer and implement the Mountain Catchment Area Act, however, this is not happening due to limited funding and resources and a general confusion around the status of the Act (Rabie & Burgers 1997; Bennett & Kruger 2015; van Wilgen et al. 2016a).

2.1.3.2 Climate and biophysical details

The Groot Winterhoek Mountains are typical of the mountains of the Cape Folded Belt of the Table Mountain Group. It consists largely of sandstone-quarzitic soils with certain areas on and near the shale-band comprising shale or partly shale-derived soils (Campbell 1983; Bands 1985; Manning & Goldblatt 2012). The sandstone-quarzitic soils support sandstone fynbos vegetation, while the two distinct bands of shale support shale-band fynbos vegetation (Campbell 1983; Mucina & Rutherford 2011; Manning & Goldblatt 2012; SANBI 2012). The vegetation of the area includes 91.2% Sandstone Fynbos, 3.8% Shale Band Vegetation, 2.9% Shale Fynbos, 0.87% Freshwater Wetlands, 0.7% Shale Renosterveld, 0.43% Alluvium Fynbos, 0.02% Sand Fynbos, 0.01% Zonal and Intrazonal Forests (Mucina & Rutherford 2011). Elevation ranges between 109 and 2063 m (mean altitude: 836 m) (DEA 2016; NGI-DEM 2016) (Figure 2.4).

The area experiences a Mediterranean-type climate with cool, wet winters and hot, dry summers (Manning & Goldblatt 2012). Mean annual rainfall is ~742 mm and ranges from 353-1776 mm (WR2012 2012). Areas above 900-1300 m receive appreciable amounts of fog and mist throughout the year. Snow occurs during the winter months in areas with altitudes greater than 1600 m (Bands 1985). The major controls on hydrology in the study area and more broadly in the region are climate, topography as well as the soil and vegetation characteristics. The winter rainfall regime means that most of the precipitation occurs when potential evapotranspiration is at or near the annual minimum. Therefore, a large proportion of winter precipitation is converted into streamflow. In contrast, summers are characterised by low precipitation, high potential evapotranspiration and conditions of low streamflow and soil moisture. In addition, the summer is characterised by increased agricultural, industrial and domestic water demands (New 1999).

2.1.3.3 Socio-economic details and major threats

The Groot Winterhoek Mountains fall within the Bergrivier and Cederberg municipalities. Although, it is mostly comprised of rocky rugged uplands and infertile soils, it is surrounded by relatively fertile and topographically subdued lowlands where the clear majority of agri-business is located. Economic activities on the lowlands is primarily and historically agricultural. However, tourism is also becoming a major contributor to the economy. The most widespread land use is dryland wheat agriculture, where irrigation permits grapes, citrus and other high-income value crops are also grown (Halpern & Meadows 2013).

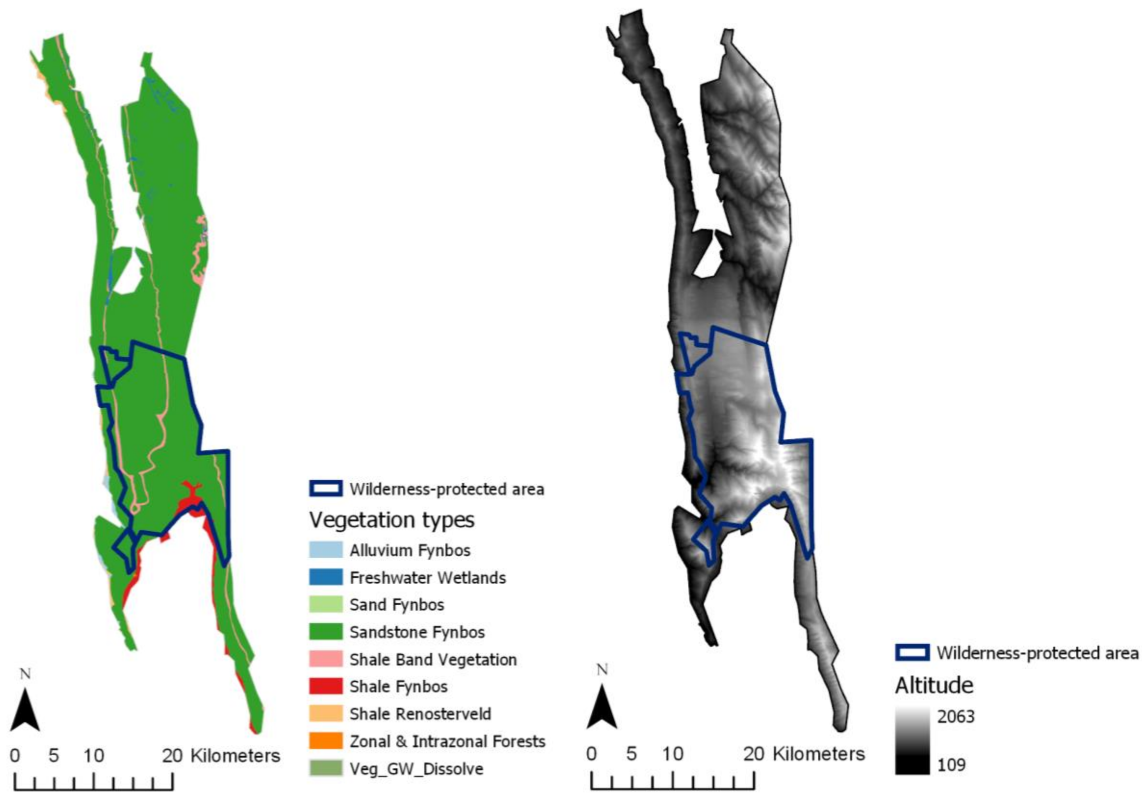


Figure 2.4 Vegetation types (map on the left) and altitude (map on the right) in the Groot Winterhoek Mountains. Also showing the boundary of the Groot Winterhoek Wilderness Area (wilderness-protected area).

Land use outside the Groot Winterhoek Wilderness Area within the Groot Winterhoek Mountains consists mostly of cultivation of indigenous plants (e.g. Proteas) for the cut flower industry, blueberry farming as well as ecotourism and private recreation. There have been no farming activities inside the wilderness-protected area since 1978. The area is currently used for recreation, although under strictly regulated and controlled conditions and it has been closed to the public since 2016 in an attempt to assist vegetation recovery from damaging fires that have occurred (CapeNature 2016a).

The key threats to the study area and surrounding lowlands include poverty, climate change and land degradation. Although, most of the land on the lowlands and in the mountains are owned by high-income or working-class income earners there is poverty in the townships with the associated socio-economic problems occurring e.g. alcoholism and drug abuse. Agriculture in the region is critical for employment and therefore any decreases in agricultural production will have impacts on farmworkers and associated value chains. Major land degradation has occurred in the lowlands in the past due to poor soil management, however despite improvements in soil management practices, biodiversity and ecosystem services are increasingly being threatened due to increases in farmland and in particular conversion to intensive agriculture (Meadows 2003; Halpern & Meadows 2013). An increase in frequent large fires on the mountains is a major threat to mountain biodiversity as well as socio-economic activities on the lowlands and mountains.

2.1.3.4 Existing conservation and other related initiatives

Given that it is impossible to place the entire region under strict regulation, CapeNature, the provincial conservation authority in the Western Cape, has a dedicated Biodiversity Stewardship Programme, which offers a range of conservation options to landowners to support and encourage responsible management on their land (Figure 2.5). In return, landowners receive appropriate benefits, which are aligned with the level of conservation under which the land is placed (Lochner et al. 2003; van Wilgen et al. 2016a).

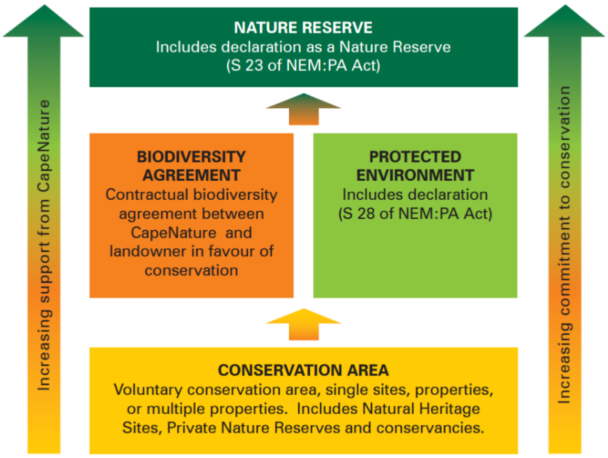


Figure 2.5 Four voluntary options available to landowners for a partnership with CapeNature. The higher the category the more incentives (benefits) and support a landowner recieves however the greater commitement required from the landowners.

There is also a range of other biodiversity mainstreaming tools that have been made available for use in land use planning and within the production sectors that are relevant to the Groot Winterhoek Mountains. These are mostly as a result of the hugely successful Cape Action for People and the Environment programme, a partnership between government and civil society formed in 2001 (see Younge & Fowkes (2003) for an overview of this planning process). An example is the Critical Biodiversity Area (terrestrial and freshwater) maps that have been developed as guidance tools by the South African National Biodiversity Institute in addition to their spatial decision support tools such as the land use decision support spatial planning and assessment tool. There has also been a range of biodiversity criteria developed and introduced into production standards associated with certain biodiversity initiatives for wild flower harvesting, potatoes, rooibos, citrus and wine (Allsopp et al. 2014).

The Greater Cederberg Biodiversity Corridor implemented by CapeNature is a project that assisted in securing private land in the Groot Winterhoek Mountains under the Biodiversity Stewardship Programme for conservation. The Greater Cederberg Biodiversity Corridor originated largely from the Cape Action for People and the Environment. The Groot Winterhoek Freshwater Stewardship Corridor Project formed part of the Greater Cederberg Biodiversity Corridor programme and included an awareness campaign focused on school learners and farm workers and the eradication of spotted bass, an invasive alien fish from the Thee Rivers (Paterson 2012).

In terms of fire management across the landscape, the Greater Cederberg Fire Protection Association is a group of landowners that prevent and manage wildland fires in the broader Cederberg area with the overall aim of achieving integrated wildfire management. This includes most but not all the landowners in the Groot Winterhoek Mountains. Landowners join the Fire Protection Association to make sure that they are compliant with the National Veld and Forest Fire Act of 1998. Membership is voluntary, but it can assist in terms of legal issues related to the spread of run-away fires. Landowners that are apart of Fire Protection Associations get access to information on managing fire risk at a landscape level and get assistance from Working on Fire pre- and post-fire season and during fires.

Working on Fire is an Expanded Public Works Programme aimed at providing work opportunities to young men and women and is funded by the national Department of Environmental Affairs. Working for Water also operates in the study area with the aim to clear alien plants on landowners' land. It is also an Expanded Publics Works Programme and has won many international awards for achieving two different goals which include both creating jobs as well as protecting biological diversity and water resources through the clearing of alien plants (Binns et al. 2001; van Wilgen & Wannenburgh 2016; Angelstam et al. 2017).

2.1.3.5 Hydrology and water resources

The study area includes mountain sub-catchments from two primary drainage catchments. These include the i) Berg, and ii) Olifants, which drain the mountains that run parallel to the West Coast (Figure 2.6). Runoff from the study area drains into the Berg and Olifants Rivers through several tributaries, including the Twenty-Four-Rivers, the Dwars River and the Ratel River. Runoff from the Berg tributaries is channelled into Voelvlei dam, which contributes to the domestic and industrial water for Cape Town via a network of reservoirs and inter basin transfer schemes known as the Western Cape Water Supply System. It also contributes more locally to towns in the Bergrivier municipality such as Porterville and provides for downstream-irrigated lands and farming communities. The Olifants is an important source of water for irrigated agriculture north of the Berg including the Citrusdal valley (Bands 1985; New 1999; River Health Programme 2004). Runoff that drains into the Olifants River is used primarily for the irrigation of citrus crops in the Citrusdal valley and for the supply of water to several large dams in the area such as Clanwilliam and Bulshoek (Bands 1985; River Health Programme 2004).

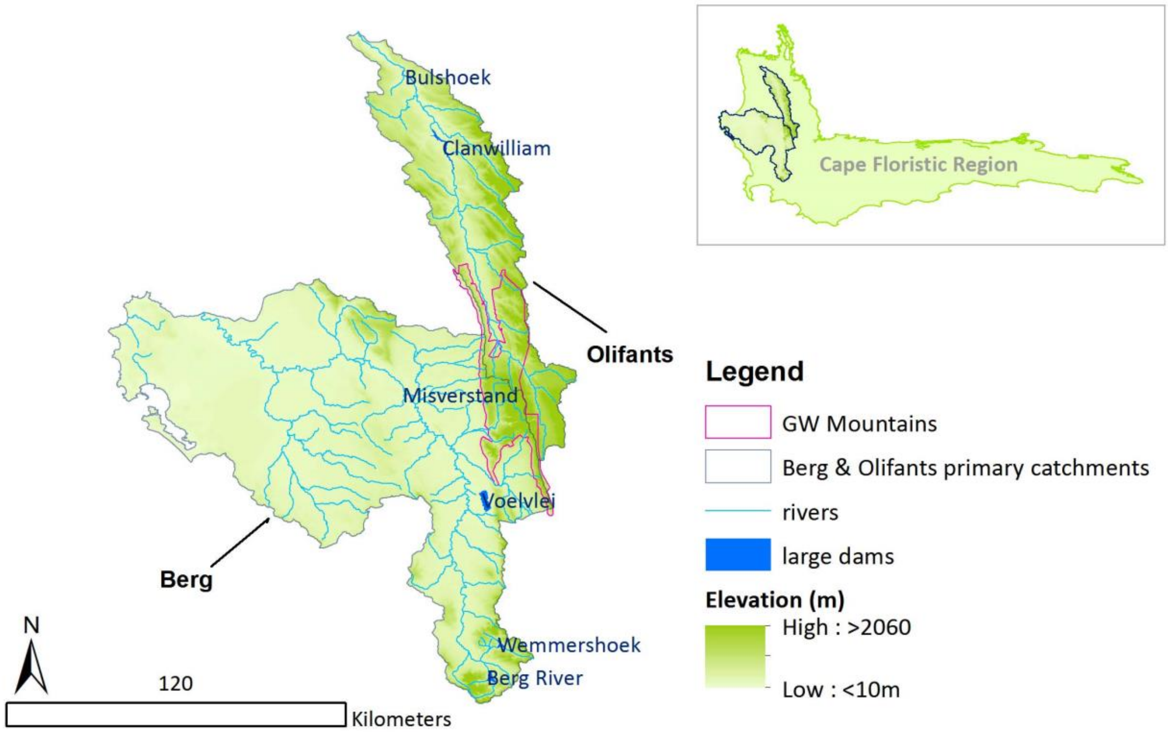


Figure 2.6 The Groot Winterhoek Mountains in relation to main primary catchments in the Cape Floristic Region, South Africa (NGI 1997; CSIR 2007; Schulze 2008b; Schulze et al. 2011; SANBI 2012; DEA 2016). Water from the Groot Winterhoek Mountains is important for the Voelvlei, Misverstand, Clanwilliam and Bulshoek dams as well as irrigated agriculture in nearby lowlands and other farming activities. Voelvlei dam is an important component of the Western Cape Water Supply System and provides drinking water for various urban centres including Cape Town. Water leaving the Groot Winterhoek is channeled into Voelvlei dam via a canal network which then links to an intricate water transfer system with other primary drainage basins in the region (River Health Programme 2004). Please refer to Figure 2.1 for the wider geographical area within which the Cape Floristic Region is apart.

2.2 Constructing the counterfactual: a pluralistic, socio-ecological approach

I constructed the counterfactual conditions for protected area impact evaluation using a pluralistic, socio-ecological approach that promotes disciplinary rigor, longitudinal perspectives, and integrated research and knowledge. Several socio-ecological system frameworks and concepts were reviewed before identifying three concepts, namely ecosystem services, socio-ecological systems and land use transitions, for incorporating into the pluralistic, socio-ecological approach used in the study. I outline the most well-known of these frameworks in Section 1.4 and provide references to papers that have reviewed additional frameworks.

Existing frameworks, such as the Social-Ecological Systems Framework (Ostrom 2009) and resilience approaches (Folke et al. 2010; Folke 2016) were limited in terms of their engagement with the existing protected area effectiveness literature, and none used the concept of counterfactual thinking (i.e. what could have been the outcome for the environment if the protected area or intervention had not been established) in their design. In addition, many lacked temporal depth and the explicit identification of where, when and how to engage with trans- and interdisciplinary inquiry. This included how the many different existing socio-ecological concepts could be used and where and when stakeholders could be involved.

The three concepts of ecosystem services, socio-ecological systems and land use transitions were identified as suitable for the study of protected area impact due to their integrated nature as well as the ease in which they could be integrated and used across disciplines and different frameworks (see Section 1.4 for more detail on this). Pluralism as an approach was adopted in this study due to its flexibility and openness. Pluralism was used to connect the concepts of ecosystem services, socio-ecological systems, and land use transitions to the literature on protected area impact evaluations, specifically the concept of counterfactual arguments. Multiple disciplines as well as data, information and knowledge sources were then drawn on to achieve these connections. A full description is provided below.

2.2.1 A pluralistic, socio-ecological approach

I started out with the typical directed acyclic diagram for protected area impact evaluations suggested by Ferraro & Hanauer (2015) and presented in Figure 1.1 in Section 1.2. Building on this starting point, I envisaged an additional nine components. Six components were operationalised in the study (see 1-6 in Table 2.1). The last three components (7-9 in Table 2.1) were not but are presented for inclusiveness and to inform future research endeavours. The methodology was developed specifically for in-depth place-based and landscape level analyses.

Through pilot interviews with landowners and local conservation and mountain management organisations I identified three important outcome variables for the mountains, namely fire, vegetation and water and one relevant mechanism effect, namely land use/cover change. I then aimed to determine the impact of the protected area on this mechanism effect and these three outcome variables using a counterfactual framing while considering socio-ecological context.

To generate counterfactual conditions, I used three main sources of information: i) repeat social surveys and in-depth interviews; ii) orthorectified aerial imagery before and after protected area establishment both inside and outside the protected area; and iii) modelling the influence of land use/cover scenarios generated using i and ii on hydrological response inside and outside the protected area both before and after protected area establishment. I used the concepts of land use transitions, ecosystem services and socio-ecological systems in the analyses to understand the influence of confounding factors. The diagram in Figure 2.7 below and related explanation in Table 2.2 were used to understand the potential dynamics captured in the red circle which encloses the concepts of land use transitions, ecosystem services and socio-ecological systems in steps 4 – 9 in Table 2.1.

I used these three sources of information (i-iii described above) as well as the three concepts to understand the impact of the protected area on vegetation, fire and water flow using a counterfactual framing i.e. contextualising change inside the protected area with change and drivers of change outside the protected area on similar terrain as well as the conditions before the protected area was established.

To generate an understanding of counterfactual conditions, I firstly identified and described change and drivers of change in land use/cover outside the protected area. To do this, I interviewed landowners situated around the protected area on similar terrain to generate an understanding of changes in land use/cover outside the protected area as well as to determine drivers of this change and the influence of these changes on vegetation, fire and water flows. The methodology used was drawn mostly from the social sciences but was broadened by using a lens of historical ecology (see Table 2.1 step 3 and 4). I fully describe this methodology and present the results of this in Section 3 of the thesis.

Therefore, I start building the counterfactual conditions in Section 3 of the thesis for the protected area impact evaluation. This includes acquiring an understanding of mechanisms and confounding factors that could mask or mimic protected area impact on changes observed inside the protected area. For example, in Section 3, I describe change and drivers of change in land use over the last four decades for an area outside the protected area of similar terrain to inside the protected area. Section 3 is critical to understand changes in land use/cover that could have occurred inside the protected area had the protected area not been established. The results of Section 3 provide counterfactual arguments for understanding the influence of protection on vegetation change in Section 4 and for understanding change in land use/cover types such as ecotourism, personal nature-based recreation, wildflower harvesting, grazing and land use management practices associated with cultivation which are not readily discernible from remote sensing imagery. Section 3 is thus directly used for determining the hydrological modelling parameters and coefficients used in counterfactual scenarios for Section 5.

In Section 4 of this thesis, I determine vegetation change ~40 years after protected area establishment and then discuss the role of the protected area in changing the mechanism (i.e. land use/cover) that could have influenced changes observed in vegetation cover. I evaluate the role of climate as a moderator of protected area impact on vegetation change and use the results on land use/cover change outside the protected area in Section 3 and in Section 5 (see below) to interrogate the influence of the protected area on vegetation cover and composition inside the protected area.

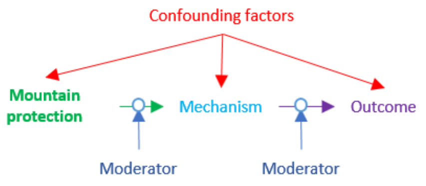
In Section 5, I draw on remote sensing to confirm changes in mechanism effects determined in Section 3 and then use hydrological modelling to consolidate the information derived from social science accounts of land use/cover change, vegetation changes observed and aerial photograph analysis into counterfactual scenarios for determining protected area impact on water flows. Specifically, I generate counterfactual scenarios that consider land use/cover prior to protection in 1949 and 1972 as well as land use/cover outside the protected area on similar terrain in 1949, 1972 and currently. I use these scenarios to model the influence of what would have occurred inside the protected if the protected area had not been established i.e. What if things remained the same? Or what if things changed in congruence with changes outside the protected area?

Hydrological model coefficients and parameters were generated using information from Section 3, an analysis of aerial imagery described in Section 5 as well as from vegetation change data generated in Section 4**.** For example, a range of variables need to be parameterised for each land use/cover type used in the hydrological model. These parameters are influenced by vegetation cover and land use management practices. Vegetation cover estimates and land use management practices for the model are drawn directly from Section 3 and Section 4 for Section 5. These are refined using additional interviews with representative stakeholders. The emphasis is on determining the influence of protection while considering the impact of changes in land use/cover inside and outside the protected area both 29 and 6 years before protection and 36 years after protected area establishment.

Table 2.1 The approach used in this study to understand the long-term impact of mountain-protection on land use and cover (mechanism), and fire, vegetation and water flows (outcome variables/processes). Components 7-9 were not operationalised in this study due to time and funding constraints.

<table>
<tr><th>The pluralistic, socio-ecological approach used in this thesis</th><th>Description</th></tr>
<tr><td>

1.



</td><td>**Starting point:** a typical protected area impact evaluation framework suggested by Ferraro & Hanauer (2015).</td></tr>
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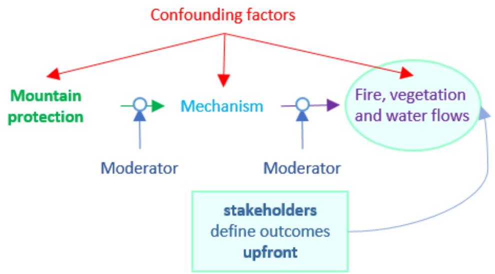
2.



</td><td>**Stakeholder initial engagement:** Stakeholders define outcome variables through pilot meetings and interviews.</td></tr>
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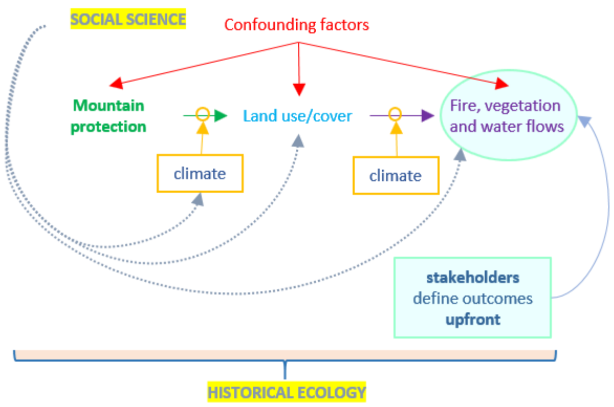
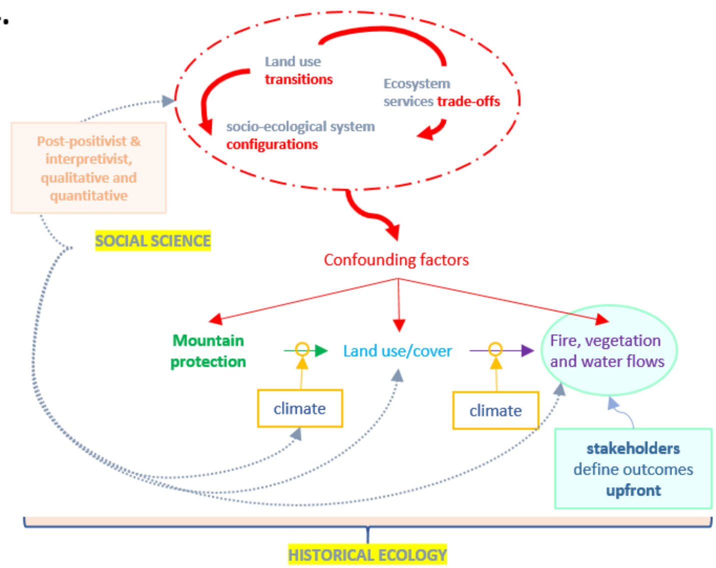
3.



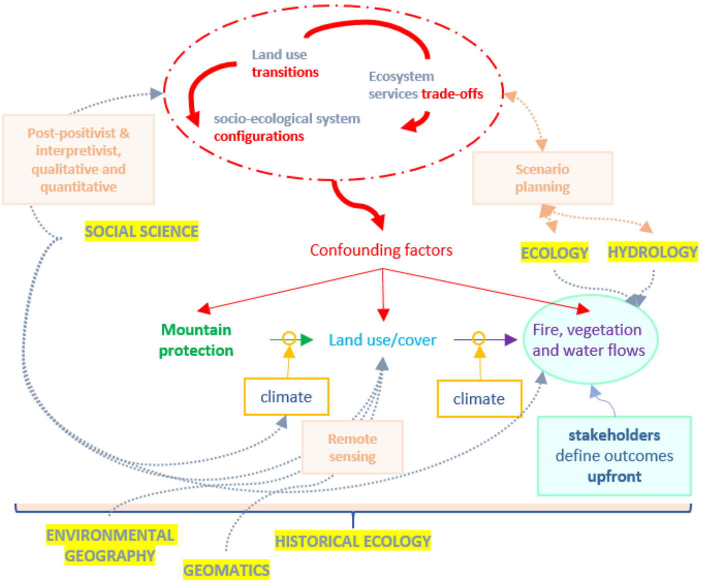
</td><td>**Stakeholder in-depth engagement:** Mechanism and moderator effects determined and the importance of the outcome variables confirmed through interviews drawing on social science and historical ecology.</td></tr>
<tr><td>

4.



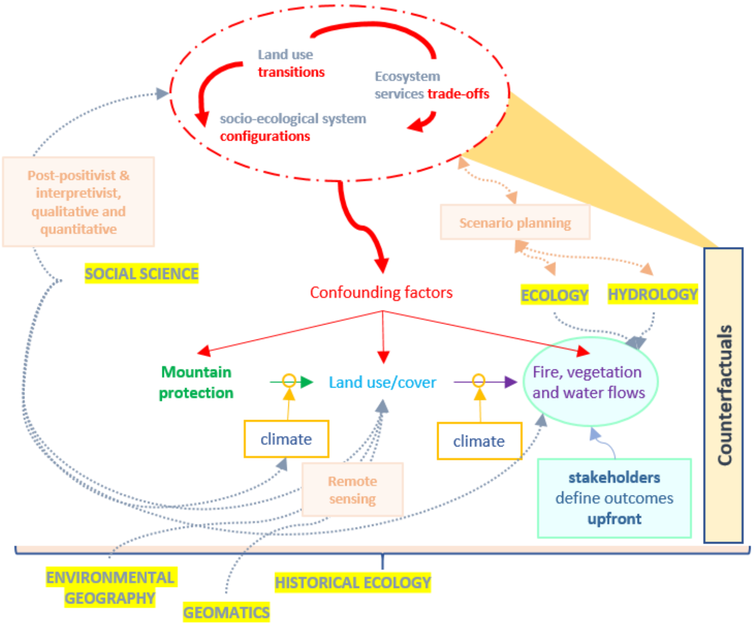
</td><td>**Evaluation of confounding factors:** Confounding factors evaluated drawing on social science and historical ecology and using the concepts of land use transitions, ecosystem service trade-offs and socio-ecological system configurations. These concepts are integrated to understand multi- and cross-scale interactions between endogenous and exogenous socio-ecological responses and socio-economic dynamics (also see section Figure 2.7 and Table 2.2).</td></tr>
</table>

5.



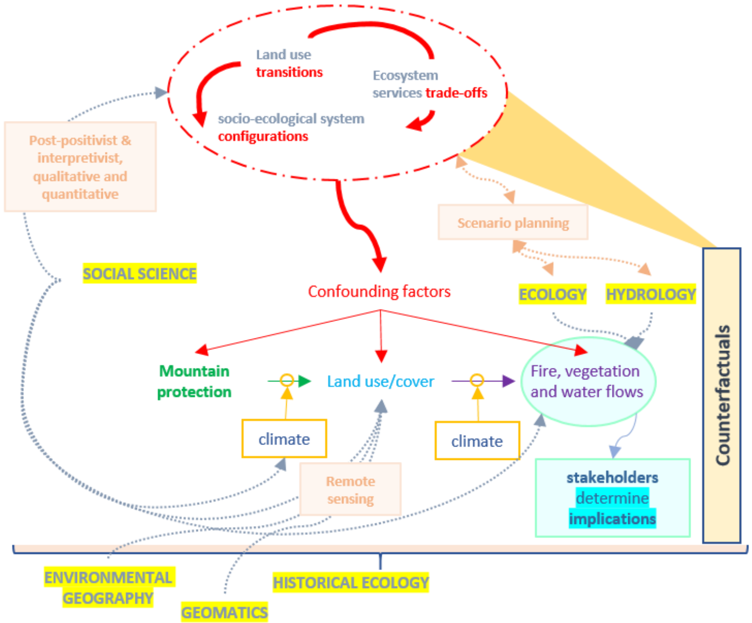
Integration of disciplines and tools to determine change in mechanism effects: Further disciplines (e.g. ecology, hydrology, environmental geography and geomatics) and tools (e.g. remote sensing and scenario planning) determined and integrated to obtain evidence on change in mechanism effects (i.e. land use/cover) and outcome variables (i.e. fire, vegetation and water flows).

6.



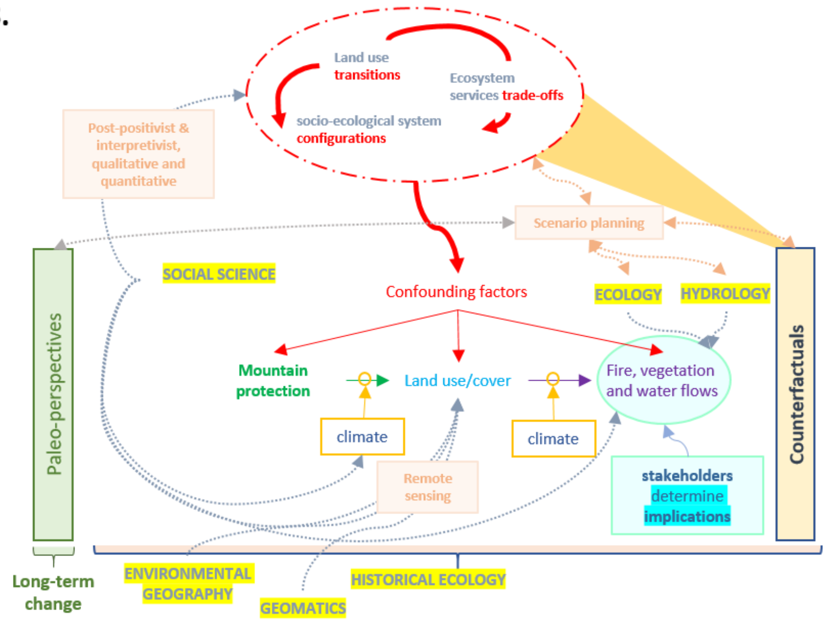
Protected area impact determined using counterfactual thinking: **Counterfactual thinking** used to contextualise mechanism effects for determining protected area impacts on outcome variables i.e. considering confounding factors and the influence of moderators.

7.



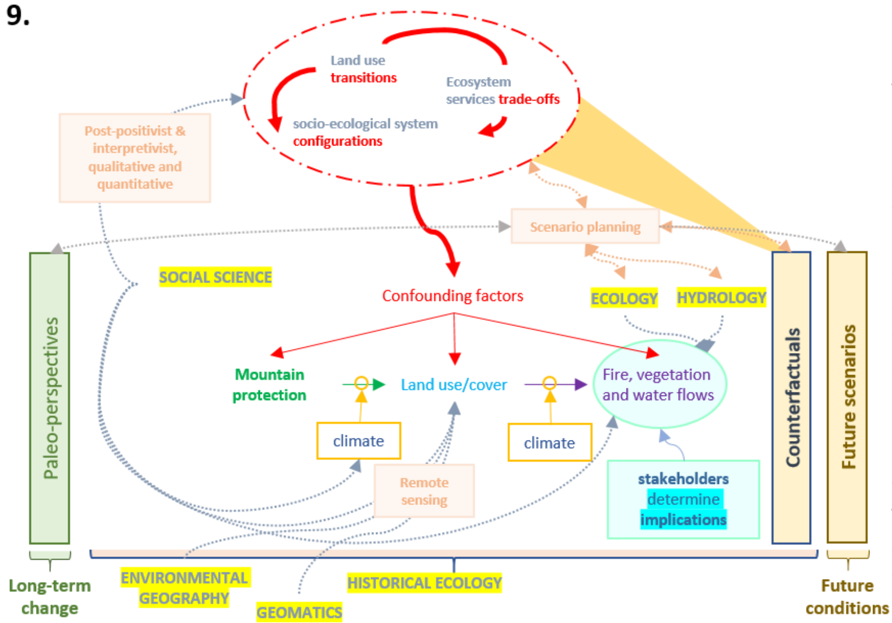
Engagement with stakeholders to determine implications of results and the necessity of paleo- or future scenario perspectives: Stakeholders determine implications of results and are engaged in the process of determining the relevance of paleo-perspective to inform future scenarios and modelling (see next two steps). This was not covered in this study but included in the framework for completeness.

8.



Longer-term change used to contextualise protected area impact: Paleo-perspectives used to determine reference points and benchmarks by drawing on paleoecology and informed through the previous step. This is not covered in this thesis but recognised within the framework for completeness. The paleo-perspectives component of the framework would require full development.

9.



Future scenarios of change for informing management: Retrospective findings and insights used to contextualise scenarios of future conditions while drawing on paleo perspectives and additional disciplines such as climate science and the literature on climate change adaptation, and tools such as scenario planning and modelling. This is not covered in this thesis but recognised within the framework for completeness. The future scenarios component of the framework would require further development.

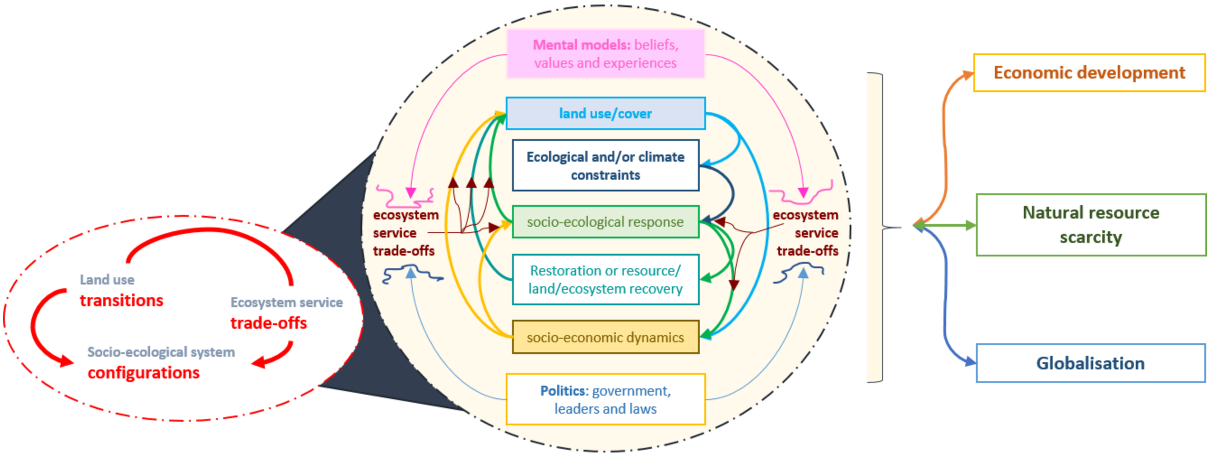


Figure 2.7 An explanatory diagram to show connections between the concepts of land use transitions, socio-ecological systems and ecosystem services in order to understand the role of confounding factors in protected area impact evaluations.

Table 2.2 A description of the potential dynamics shown in Figure 2.7 and which connect the concepts of land use transitions, ecosystem services and socio-ecological systems for understanding the role of confounding factors on conservation.

List of potential socio-ecological and socio-economic dynamics
• Land use/cover can be influenced by ecological and/or climate constraints and/or socio-economic dynamics. • Ecological and/or climate constraints can result in a local-level socio-ecological response which can result in i) resource, land or ecosystem recovery or restoration and/or ii) an alternative land use/cover type. • Local level or exogenous socio-economic dynamics can result in i) resource, land or ecosystem recovery or restoration and/or ii) an alternative land use/cover type. • Local socio-ecological responses to ecological and/or climate constraints can be further influenced or moderated by exogenous socio-economic dynamics that result in i) resource, land or ecosystem recovery or restoration and/or ii) an alternative land use/cover type. • Ecosystem service trade-offs are an essential and integral component of socio-ecological responses to i) ecological and/or climate constraints; and/or ii) local level or exogenous socio-economic dynamics. • Ecosystem service trade-offs influence whether i) resource, land or ecosystem recovery or restoration and/or ii) an alternative land use/cover type occurs. • In a spatial context, ecosystem service trade-offs influence how much land is used for certain land use/cover types, restored or allowed to recover to a natural (Lambin & Meyfroidt 2010) or novel ecosystem state (Hobbs et al. 2014). • Ecosystem service trade-offs are influenced by both the current politics at the time but also individual mental models. Politics and mental models can either agree or oppose each other. This influences the type of ecosystem service trade-offs made at a local level (Biggs et al. 2015). • Trade-offs result in ecosystem service interactions i.e. when changes in one-ecosystem service results in changes in another ecosystem service (Bürgi et al. 2015). • The overall resulting socio-ecological landscape comprising land use/cover types and available ecosystem services is the socio-ecological system configuration (Biggs et al. 2015). • The overarching pathways of change that can play a role in driving local and exogenous socio-economic dynamics and socio-ecological responses are globalisation, natural resource scarcity and economic development (Lambin & Meyfroidt 2010).

Section 3. Subsistence grazing to nature-based recreation: Land use transitions and drivers in a mountain case study of a global biodiversity hotspot

3.1 Introduction

This section of my thesis focuses on change in private land use over the last four decades in the Groot Winterhoek Mountains (see 2.1.3) and relates change to relevant national and local conservation policies including the Mountain Catchment Areas Act 63 of 1970 and the National Veld and Forest Fire Act 101 of 1998 (see section 2.1.2.1, 2.1.2.2 and 2.1.3.4). It furthermore captures the views and values of landowners on the formal protection of mountain land in the Groot Winterhoek Wilderness Area managed by CapeNature, the regional conservation organisation for the Western Cape in South Africa. Uniquely, I repeat a structured questionnaire originally conducted ~40 years ago.

I use mixed social science methods to understand trade-offs in the biophysical and social aspects of several land use types including ecotourism, personal nature-based recreation and wildflower harvesting, typically excluded from land use change studies which primarily rely on information derived from remotely sensed data (DeFries et al. 2004). Drawing on the concepts of socio-ecological systems, ecosystem services and land use transitions, the section contextualises private land use change and determines current landowner views and values of conservation. In doing so, I construct counterfactual conditions for contextualising protected area impact in Section 4 and for developing counterfactual scenarios in Section 5. Specifically, by understanding change outside the protected area and the main drivers of these changes, one can determine whether changes seen inside the protected area would have occurred in the absence of the protected area.

Given that abandonment of farmland in mountains has occurred extensively over the last 50 years, the aim of this section is to advance the theoretical understanding of the causal mechanisms of land use change in mountainous regions and the role of cultural ecosystem services in land use trade-offs associated with farmland abandonment. The study focuses on understanding the following aspects:

1) The importance of cultural ecosystem services (e.g. personal nature-based recreation and ecotourism) in ecosystem service trade-offs associated with land use change and farmland abandonment in mountain regions

2) The role of endogenous socio-ecological feedbacks versus broader socio-economic dynamics in determining land use transitions in mountain ecosystems

3) Associations between land use types and views and values of conservation of mountain land and the implications for determining appropriate leverage points for garnering societal support for protected areas

3.2 Methods

3.2.1 Case study

Change in land use was investigated in the Groot Winterhoek Mountains surrounding the Groot Winterhoek Wilderness Area (Figure 3.1 c). Specifically, this research focused on 540 km^2 of privately owned mountain land fully surrounded by the outer boundary of the Groot Winterhoek Mountain Catchment Area administration boundary according to the South African National Department of Environment National Protected Areas Database (DEA 2016) (Figure 3.1 a-b).

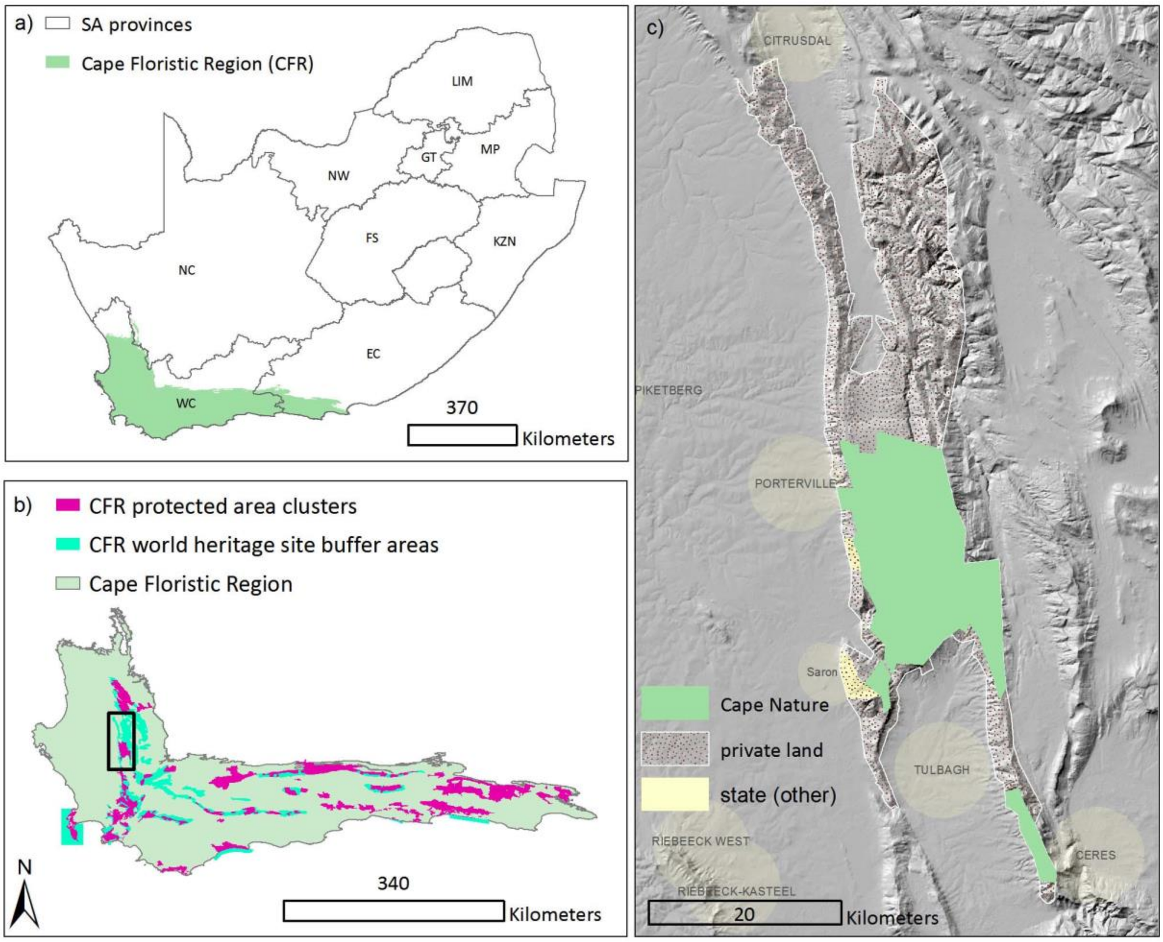


Figure 3.1 a-b) The Cape Floristic Region in South Africa indicating the 13 World Heritage Site protected area clusters and buffer zones (SANBI 2012; MDB 2013; DEA 2016); and c) the case study area indicating the outer boundary of the Groot Winterhoek Mountain Catchment Area in which private landowners were surveyed in 1978 (Bands 1985) and resurveyed in 2016. The Cape Nature managed Groot Winterhoek Wilderness Area and Nature Reserves and other state land is shown in yellow and green (SUDEM 2014; DEA 2016; SG 2016).

3.2.2 Mixed methods approach

Interviews with private landowners in the Groot Winterhoek Mountain Catchment Area were conducted to elicit both closed-structured responses as well as open-ended and in-depth responses. A concurrent triangulation mixed method design was followed to integrate quantitative and qualitative information (Creswell et al. 2003; Castro et al. 2010; Creswell

2013). This was contextualised within an interpretive approach (Mottier 2005; Guest et al. 2012). Specifically, the method included the following sequential stages: i) interview and questionnaire design (see final questionnaire in Appendix 3.1); ii) sample selection; iii) interviews; iv) data transcription and processing; v) qualitative analyses; vi) quantitative and statistical analyses; and vii) integration of quantitative and qualitative information. Equal consideration was given to quantitative and qualitative information throughout the mixed methods process.

3.2.2.1 Interview questions

A structured questionnaire developed and conducted by Bands (1985) in 1978 with 83 landowners who owned properties covering 514 km^2 of the privately owned mountain catchment, depicted in Figure 3.1 c, was repeated in 2016 (38 years later) with 60 present day landowners owning 110 farm portions randomly selected from the same area but with properties covering 441 km^2.

The original survey conducted in 1978 focused on land use type and area in the mountain catchment. It included four sections that captured whether a landowner used their specific mountain property for grazing, cultivation, private recreation (e.g. family or friends), ecotourism (recreational use for economic reasons) and/or wildflower harvesting, as well as the size of the property that was utilised for each land use type. These original sections were included in the repeated questionnaire in 2016. The original questionnaire, however, was further expanded to obtain additional quantitative and qualitative information using insights gained through interviews with individuals from relevant organisations working in the local area, including representatives from the Greater Cederberg and Winelands Fire Protection Associations, Cape Nature (Porterville, Tulbagh and Wolseley offices), the Tulbagh Agricultural Committee, three local landowners and two past landowners from the area. This process was followed to ensure the questionnaire would capture information that was of relevance to both local landowners but also to organisations associated with conservation and mountain management.

Structured, quantitative, focused sections were added to the questionnaire to capture landowner views on the regulation and protection of mountain land for conservation and protection of water. This included landowner views on the type of value (intrinsic-moral, economic non-use and use value) placed on the protection of a portion of the mountains in the Groot Winterhoek Wilderness Area. Economic value was informed by standard economic theory (i.e. anthropocentric view) and included use and non-use value of conservation. Intrinsic value was based on ethical and philosophical grounds (i.e. biocentric view) which suggests that species have a right to exist and that the most fundamental source of value should not be limited to perceptions that form the basis of economic analysis (Harris & Roach 2013). Questions on current land use practices, specific descriptor landowner characteristics as well as the levels of awareness and influence of the Mountain Catchment Areas Act 63 of 1970, the National Veld and Forest Fire Act 101 of 1998 and local conservation programmes were also included. Open-ended, in-depth interview style questions were added to the 2016 study to obtain a comprehensive understanding of the drivers of current land use and management in the catchment as well as to attain information on perceived change in the mountain catchment in terms of economic value and environmental sustainability as well as the drivers of this change (Table 3.1 and Appendix 3.1).

Table 3.1 Topics added to a questionnaire originally conducted by Bands (1985) in 1978 and repeated with landowners in 2016.

Quantitative structured sections (closed-ended questions)
• Value (intrinsic-moral and economic) placed on the formal protection of the Groot Winterhoek Wilderness Area
• Views on state regulation and conservation in the mountain catchment (including the formal protection of the Groot Winterhoek Wilderness Area)
• Awareness of relevant local conservation programmes including the Greater Cederberg Biodiversity Corridor and the Groot Winterhoek Freshwater Stewardship Corridor
• The awareness and influence of mountain and fire legislation including the Mountain Catchment Areas Act of 1970 and the National Veld and Forest Fire Act of 1998
• Further details on current land use and management not included in the past survey with a focus on fire usage and alien plant clearing
Qualitative sections (open-ended questions and eliciting in-depth responses)
• Past land use and coverage of alien plants (if known)
• Views on perceived change and drivers of change in the mountain catchment in terms of economic value and environmental sustainability (as far back as a landowner could recall)

3.2.2.2 Sample selection and interview procedure

An interview with the original surveyor (Bands 1985) was also conducted in 2015 to obtain insight on the original questions used and the survey process, which was followed in 1978. Property parcel boundaries and names were obtained from South Africa's Chief Surveyor General (SG 2016). One hundred and thirty farm portions were randomly selected from the 152 farm portions covering the study area with the aim of achieving a reasonable sample size of landowners given that ownership can include more than one farm portion. Of this selection, seven landowners (owning seven farm portions) requested to be omitted from the study due to time constraints or disinterest and 11 landowners (owning 14 farm portions) provided no response or were unreachable due to incorrect contact details. In total 55 interviews were conducted including 53 face-to-face and two telephonic interviews. Interviews lasted between 1 and 2 hours on average. Five landowners were unable to do interviews and requested to complete a digital version of the questionnaire. Follow-up phone calls were undertaken to clarify uncertain or unanswered questions in the digital questionnaire responses.

3.2.2.3 Quantitative and statistical analyses

Table 3.2 describes landowner characteristics and property characteristics (including land use) and related response variables associated with values, views, awareness levels and land use practices used for quantitative analyses. Additional variables extracted to contextualise quantitative results include type of private recreation and ecotourism activities, views on current fire management, and views on controlled burning (Appendix 3.1, question 6.1.4, 7.1.1, 9.4). Statistical analyses were performed in R (R Core Team 2016).

3.2.2.3.1 Land use, views, values and awareness of conservation and state regulation

Pearson's chi square tests of equal proportions were used to compare observed proportions of respondents for each land use between 1978 and 2016[1]. To determine if interviewee responses were equally distributed across question multiple choice categories, the Pearson's Chi-square Goodness of fit test was used[2]. Expected frequencies were dependent on the number of categories per question. For example, expected frequencies for a binary question, e.g. yes/no, would be 0.5 and 0.5, whereas for a question with five categories expected frequencies would be 0.2, 0.2, 0.2, 0.2, and 0.2. Therefore, the null hypothesis for each test per question is: there is no significant difference between the observed and expected response frequencies across question categories. While the alternative hypothesis is: there is a significant difference between the observed and expected response across question categories. Pairwise post hoc chi square tests with BH p-value adjustments were applied to questions with multiple categories for multiple comparisons[3] (Benjamini & Hochberg 1995; Benjamini & Yekutieli 2001). Confidence intervals were generated for visual comparisons using nonparametric bootstrapping without assuming normality[4].

3.2.2.3.2 Relationship between response variables and landowner as well as property characteristics

Associations between current landowner and property characteristics (including land use) and each response variable identified in Table 3.2 were determined using focused principle component analysis[5]. Multiple binary logistic regression was used to quantify and further evaluate the strengths of associations between dependent variables and associated predictor variables (based on the results from the focused principle component analysis)[6]. Situations of multicollinearity were assessed using the results of the focused principle component (Falissard 1999; Fox & Weisberg 2011). Standard errors were checked, and correlated predictor variables were assessed using Variance Inflation Factors[7]. Differences between the model and observed data and overall model fit for each response variable were tested using the residual deviance and Pearson's goodness of fit tests and Bayes Information Criterion[8].

[1] R package: stats; function: prop.test
[2] R package: stats, functions: chiq.test
[3] R package: stats, fifer; functions: p.adjust, chisq.post.hoc (Fife 2014)
[4] R package: ggplot2, hmisc; functions: mean.cl.boot, smean.cl.boot (Wickham 2009; Harrell & Dupont 2015)
[5] R package: psy, stats; functions: fpca, cor (Falissard 2012)
[6] R package: stats; functions: glm
[7] R package: stats, car; functions: glm, vif (Fox & Weisberg 2011; Fox 2015)
[8] Rstats package: stats; glmulti; functions: anova, pchisq, glmulti (Calcagno & de Mazancourt 2010; Calcagno 2013)

Table 3.2 Landowner and property characteristics, and response variables including landowner values and views on protection and state regulation, awareness of local conservation programmes, as well as mountain and fire policy, and land use practices related to burning and alien clearing.

Data type	Details	Source
Predictor variables		
Private landowner characteristics	Number of years owned or lived on property [own.years: no.]	Interview with landowners
	Whether landowners live on the mountain property [live: Y/N]	
	Groot Winterhoek Wilderness Area visitation by landowners [GW.visit: Y/N]	
	Landowner conservancy member [conservancy: Y/N]	
Private property characteristics	Distance to the Groot Winterhoek Wilderness Area [distance.GW.m: meters]	Surveyor general cadastres (SG 2016), Department of Environmental Affairs Protected Area database (DEA 2016)[9].
	Distance to nearest Protected Area including the GW Wilderness Area [nearest.PA: meters]	
	Land use	Interview with landowners
	Ecotourism [ecotourism: Y/N]	
	Personal nature-based recreation [private.recreation: Y/N]	
	Wild plant harvesting [harvesting: Y/N]	
	Grazing [grazing: Y/N]	
	Cultivation [cultivation: Y/N]	
Response variables		
Values placed on the Groot Winterhoek Wilderness Area	Intrinsic-moral values i.e. ethical and philosophical reasons e.g. *plants, animals and ecosystems have a safe place to exist in the Wilderness Area* [intrinsic.moral: Y/N]	Interview with landowners
	Existence (non-use) values e.g. *I feel good knowing the wilderness is there for protecting plants, animals and ecosystems* [existence: Y/N]	
	Bequest and option (non-use) value e.g. for future generations [bequest.option: Y/N]	
	Soil erosion control (indirect use) value [erosion.control: Y/N]	
	Water supply (indirect use) value [water: Y/N]	
	Hiking, relaxing and recreation (direct use) value [recreation: Y/N]	
	Spiritual or cultural experiences/reasons (direct use) [spiritual: Y/N]	
Views on protection and state regulation	Merit in the protection of the Groot Winterhoek Wilderness Area [GW.merit.protect: Y/N]	
	Merit in state regulation of privately owned mountainous land [state.reg Y/N]	
Awareness of local conservation programmes and policy	Local conservation programme awareness: Greater Cederberg Biodiversity Corridor or Groot Winterhoek Freshwater Stewardship corridor [aware.conservation.progs: Y/N]	
	National Veld and Forest Fire Act awareness and influence [NVFFA.a/i: Y/N]	
	Mountain Catchment Areas Act awareness and influence [MCA.a/i: Y/N]	
Land use practices	Burn [burn: Y/N]	
	Clear alien plants [aliens.clear: Y/N]	

[9] ArcGIS tool: near table analysis (ESRI 2015)

3.2.2.4 Qualitative analyses

Qualitative analyses followed an integrated interpretive and positivist approach (Mottier 2005; Guest et al. 2012) and was performed with assistance from NVivo qualitative data analysis software (NVivo 2012). The analysis procedure used key elements from interpretivism supported by a number of quantitative and strictly, positivist techniques (Guest et al. 2012). The primary aim of this approach was to present a narrative of open-ended responses obtained through interviews with landowners. This approach also served to connect qualitative narratives to quantitative land use change results and helped to contextualise the study's findings in a conceptual model while drawing on the concept of land use transitions and associated explanatory frameworks (DeFries et al. 2004; Lambin & Meyfroidt 2010).

3.2.2.4.1 Thematic narratives of change and drivers of change

Transcripts were coded to respective questions per source item (i.e. individual interviewed) and quantitative information was imported for attribution purposes. Detailed reading was conducted of all open-ended responses focused on past land use and alien coverage as well as additional information provided by interviewees to close-ended questions. This was further cross-checked against quantitative attributes to gain an impression of the content. This approach also helped to generate ideas regarding coding categories and to identify final thematic narratives related to land use, management, drivers, and effects of changes in the catchment. Additional preliminary data analyses included using descriptive frequency analyses to examine the distributional properties of specific themes within responses. This included word frequencies followed by text search queries and word trees to expand on ideas and to formulate initial codes.

Specific qualitative analysis focused on the two open-ended sections of the questionnaire (Table 3.1). An inductive approach was used to construct the initial codes, which related to the section on drivers and effects of change. An explicit coding phase was then conducted where all data were coded to thematic categories. Codes were then grouped further and reorganised based on a second coding stage. Final coding resulted in fifteen themes: i) fire; ii) money; iii) water; iv) high intensity agriculture; v) recreation and tourism; vi) conservation; vii) aliens; viii) government; ix) market; x) awareness; xi) wildlife; xii) vegetation; xiii) lowlands; xiv) harvesting; and xv) grazing (Figure 3.2 a-b). Narratives were developed for each theme drawing on word frequency analyses integrated with multiple text search queries, word trees and an interpretive approach.

The word clouds aided a systematic approach to generating narratives in addition to the detailed reading described in the paragraph above. Specifically, the most frequent words within each coded theme across all 60 transcripts were analysed, firstly, by visualising them in a word cloud linked to a table of the percentage occurrence across transcripts and within transcripts. Secondly, specific text search queries were conducted on the 30 most frequently occurring words per theme. The information, i.e. sentences and paragraphs containing these words, connected to the 30 most frequent words per theme were used and summarised to generate the narratives for each theme. Additional themes were not required for information transcribed for open-ended sections on land use and past alien coverage. Following the approach adopted for other sections, the narrative was developed under the two original themes of land use and alien plant coverage using descriptive frequency analyses as described above.

Although qualitative analyses and resulting thematic narratives focused specifically on the two open-ended sections of the questionnaire, which were aimed to elicit in-depth responses, narratives were also informed by the entire interview process and experience with landowners. This is recognised as a key construct of qualitative research by certain authors and is inherent in the interpretive approach. Quantitative aspects, which were used to assist the qualitative interpretive approach, are presented to provide context to narratives in the form of word and thematic coding and reference frequencies and word clouds.

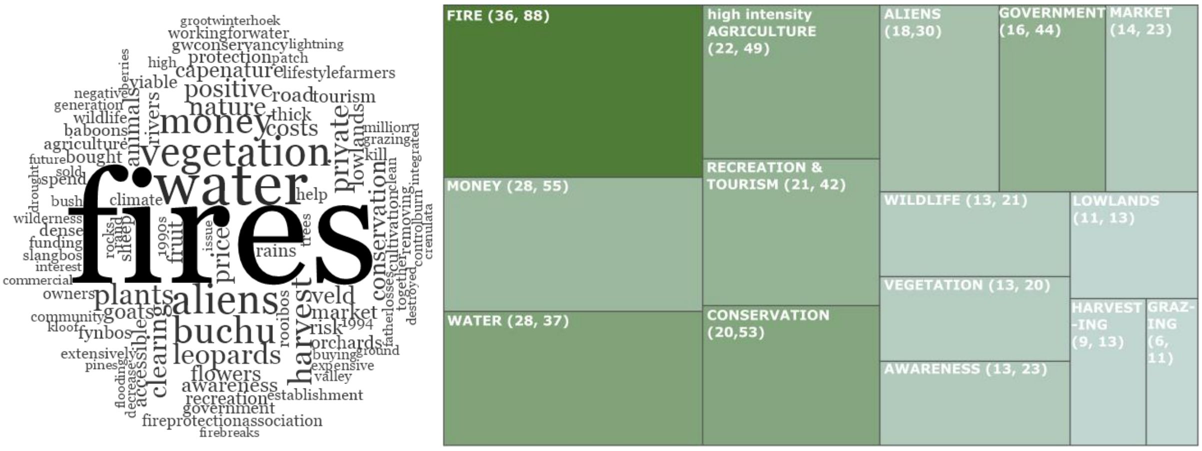


Figure 3.2 a) A word cloud indicating frequent words in responses related to perceived change and drivers of change in the mountain catchment; and b) A hierarchy chart of themes showing the 15 final themes with the size of each square and the colour an indication of the number of source items and references coded respectively. The specific number of source items and references are also shown in brackets after each theme label (total source items coded = 53, total references = 522).

3.2.2.4.2. Integrated model of socio-ecological change: land use transitions and explanatory frameworks

Integration comprised linking quantitative and qualitative information into a conceptual model of change for the catchment while contextualising and cross-checking results with recognised pathways of land use transitions and associated explanatory frameworks. Specifically, themes were condensed into several local socio-economic or socio-ecological explanations of land use change in the catchment and related ecological responses as perceived and described by landowners. These were connected to quantitative results of land use in a conceptual diagram using an interpretive approach (Mottier 2005; Guest et al. 2012). The conceptual diagram was expanded by cross-checking results with existing explanatory frameworks for general land use transitions drawing from Lambin et al. (2001; 2010), and Foley et al. (2005). These were also used to summarise the main pathways of change for the catchment and to contextualise perceptions of landowners into major causal mechanisms including socio-economic dynamics and socio-ecological feedbacks (Lambin & Meyfroidt 2010). Historical academic accounts (Wicht 1943; Ackerman 1972, 1976, 1979; Rabie 1974), government reports (Ross & Tempel 1961; Ross 1963) as well as current literature related to the influence of conservation policies on the Cape Floristic Region (Rabie & Burgers 1997; Bennett & Kruger 2015; van Wilgen et al. 2016a) were used to further interpret the land use transition pathways identified in this study.

3.3 Results

3.3.1 Questionnaire quantitative results

3.3.1.1 Land use change over the last four decades (1978 – 2016)

Private recreation (number of respondents: χ^2 = 68.16, df = 1, p<0.0001; area: χ^2 = 9520.5, df=1, p<0.0001) and ecotourism (number of respondents: χ^2 = 13.54, df = 1, p<0.001; area: χ^2 = 4542.5, df=1, p<0.0001) increased in the mountain catchment since 1978 (Figure 3.3). In contrast, grazing declined (number of respondents: χ^2 = 7.01, df = 1, p = 0.004; area: χ^2 = 9210.5, df = 1, p<0.0001). While the number of landowners using their land for cultivation increased (χ^2 = 8.29, df = 1, p = 0.0019), the area of land under cultivation remained the same (χ^2 = 0.33, df = 1, p = 0.56). There was a reduction in the area being harvested for wild plants (χ^2 = 2482.6, df = 1, p < 0.0001) but there was no change in the number of landowners harvesting wild plants (χ^2 = 0.55, df = 1, p = 0.45) (Figure 3.3). Personal nature-based, and ecotourism recreational activities included hiking, getting away from the city and relaxing in nature, fynbos, and the wilderness experience (Figure 3.4).

3.3.1.2 Landowner views on protection and reasons for valuing the Groot Winterhoek Wilderness Area

Most landowners indicated that they valued the strict protection of the Groot Winterhoek Wilderness Area for non-use existence reasons. This was followed by intrinsic-moral reasons for valuing protection as well as bequest or option non-use economic reasons i.e. for future generations. The only use value that scored equivalently to these non-use and intrinsic-moral values was the indirect economic use value of water quality and supply. Direct use values centred on nature-based recreation inside the Groot Winterhoek Wilderness Area and the indirect use value of erosion control scored lower with the direct use value centred on spiritual/cultural experiences inside the Groot Winterhoek Wilderness Area being of least importance. Only 5% of landowners indicated that they placed no economic or intrinsic value on mountain protection (Figure 3.5).

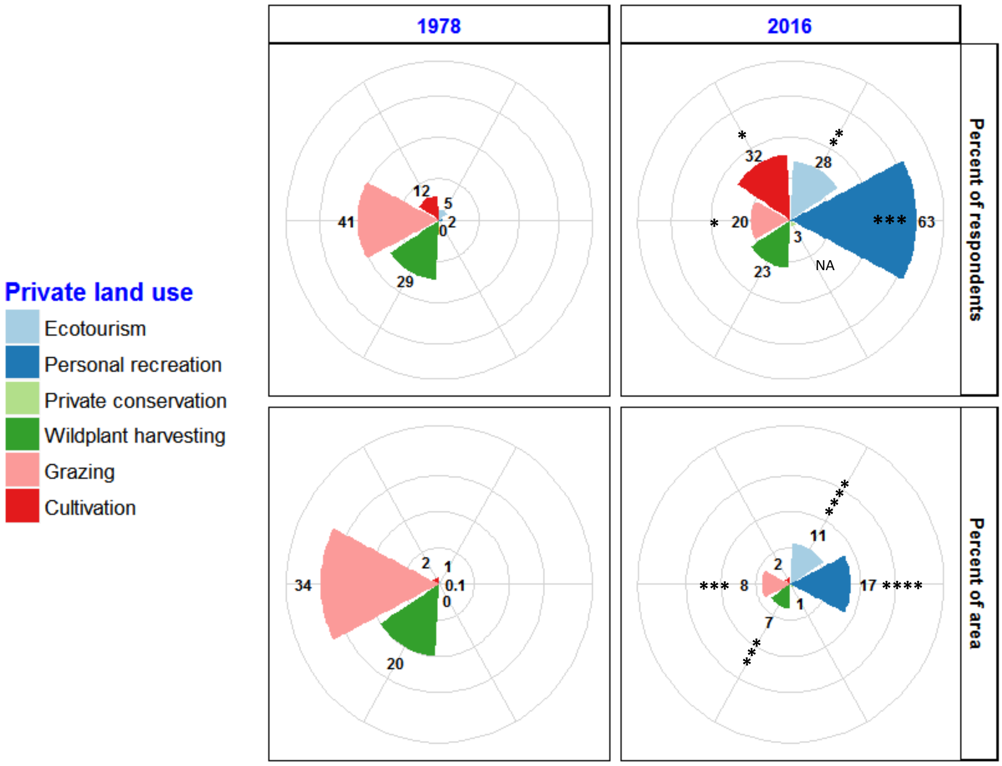


Figure 3.3 Rose diagrams of private land use change since 1978 in the Groot Winterhoek Mountains. Asterisks provide an indication of p-values (*p<0.05; ** p<0.0001; *** p<0.0001) see exact values in text above.

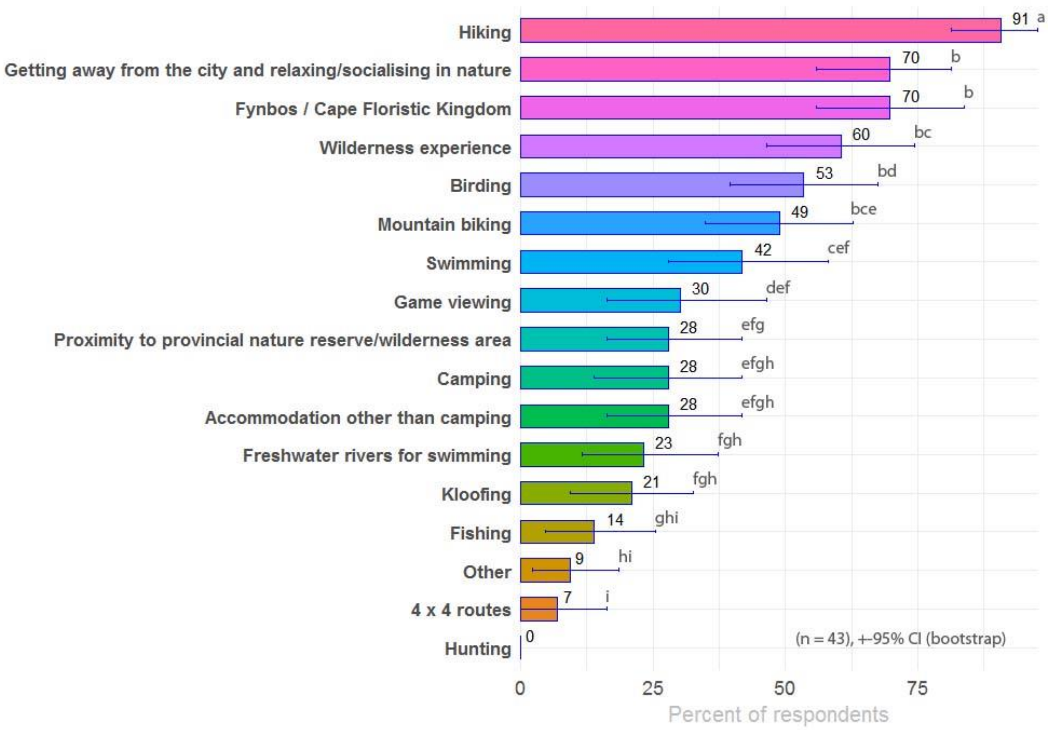


Figure 3.4 Distribution of privately conducted nature-based recreational and ecotourism activities conducted in privately owned mountain properties ($\chi2$ = 191.19, df = 16, p<0.0001). Letters a-i indicate associations between categories from multiple $\chi2$ post hoc comparisons tests (p-values <0.05) while 95% bootstrapped confidence intervals are also shown for visual comparisons.

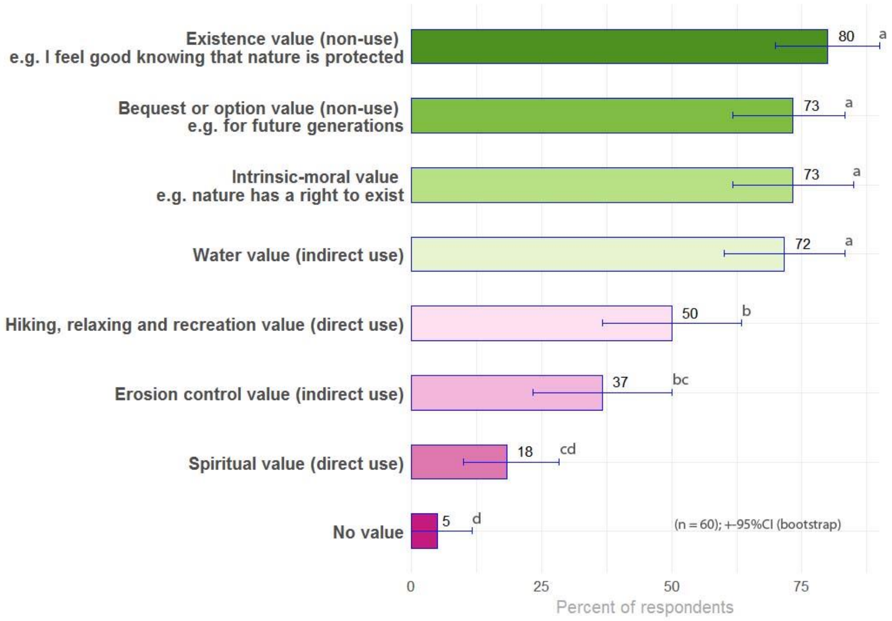


Figure 3.5 Distribution of non-use, use (direct and indirect) and intrinsic-moral values placed on the Groot Winterhoek Wilderness Area by neighbouring landowners owning land in the mountain catchment (χ^2 = 135.78, df=7, p<0.0001). Letters a-d indicate associations between categories from multiple $\chi 2$ post hoc comparisons tests (p-values <0.05) while 95% bootstrapped confidence intervals are shown for visual comparisons.

3.3.1.3 Landowner awareness and views on state regulation of private land

There was higher awareness and influence levels of the National Veld and Forest Fire Act in comparison to the Mountain Catchment Areas Act (χ^2 = 29.902; df=1; p <0.0001; χ^2 = 43.982; df=1; p <0.0001, respectively for awareness and influence) (Figure 3.6). While awareness of the National Veld and Forest Fire Act did not differ from the influence of the Act (χ^2 = 3.33; df=1; p = 0.067), awareness of the Mountain Catchment Areas Act was higher than influence (χ^2 = 9.78; df=1; p = 0.0017). This is indicative that awareness of the Mountain Catchment Areas Act was not directly relevant to management and decisions on private property. In contrast, most landowners that were aware of the National Veld and Forest Fire Act were also influenced by the Act.

Despite being highly influential the National Veld and Forest Fire Act was also highly controversial with 60% of landowners indicating that they would like changes to the current approach to fire legislation and management for privately owned mountain land (χ^2 = 38.4, df = 2, p-value < 0.001) (Figure 3.7 A). Over 80% of landowners indicated that controlled burning could be a relatively effective solution to managing fire in the mountains (χ^2 = 232.32, df = 5, p-value < 0.001) (Figure 3.7 B).

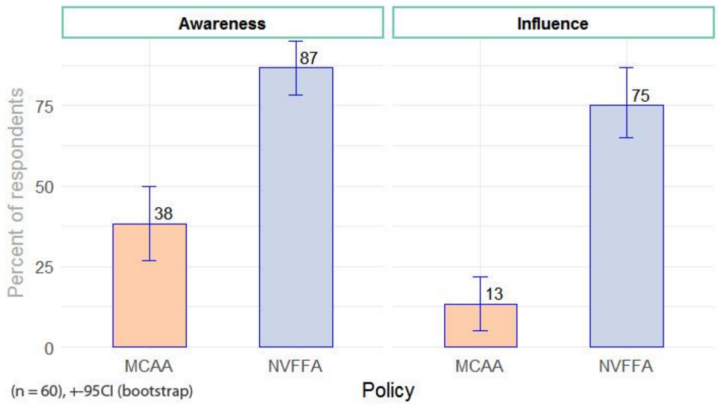


Figure 3.6 Awareness and influence of mountain and fire policy relevant to privately owned land in the Groot Winterhoek Mountain Catchment (MCAA = Mountain Catchment Areas Act; NVFFA = National Veld and Forest Fire Act). Percentages are shown with 95% bootstrapped confidence intervals for visual comparisons.

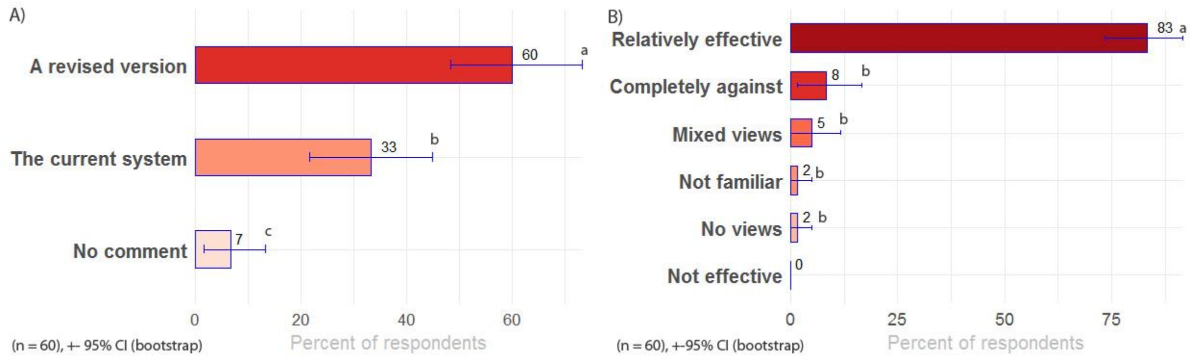


Figure 3.7 Landowner views on A) current fire management and legislation in the mountains; and B) controlled burning. "*Current system*" in (a) refers to policies related to the National Veld and Forest Fire Act of 1998 with support from the Fire Protection Association. Letters indicate associations between categories from multiple $\chi 2$ post hoc comparisons tests (p-values <0.05) while 95% bootstrapped confidence intervals are shown for visual comparisons.

3.3.1.4 Landowner views on protection and state regulation for conservation

Despite different levels of awareness between mountain protection and fire policy, 78% of landowners indicated that there was merit in the state regulating privately owned land in mountain areas for conservation and water supply benefits (χ^2 = 35.267; df = 1; p<0.0001). However, most of these landowners (87%) indicated that regulation should be conducted with effective collaboration and communication with landowners and 30% indicated that regulation should not affect the economic viability of the land. In terms of strict formal protection, 88% of landowners indicated that there is merit in the protection of the Groot Winterhoek Wilderness Area as a formal reserve for the protection of water supply and conservation (χ^2 = 57.76; df = 1; p<0.0001).

3.3.1.5 Current fire and alien land use practices, and awareness of local conservation programmes

Approximately 21% of landowners indicated that they were carrying out some sort of controlled burning programme or were actively burning their mountain land for fire management (χ^2 = 19.267; df = 1; p<0.0001). This is in comparison to 71% (χ^2 = 30.134; df = 1; p<0.0001) of landowners who indicated that they actively managed or cleared alien plants (χ^2 = 11.267; df = 1; p<0.001). Approximately half the landowners interviewed were aware of recent local conservation programmes (χ^2 = 0.6; df = 1; p<0.436).

3.3.1.6 General landowner characteristics

On average landowners had owned or lived near their property for 22 years (range: 1 to 71 years). Only 11% of landowners lived on their mountain property for the full year. Twenty-one percent of landowners interviewed formed part of a landowner conservancy recognised as a voluntary conservation area which was originally established in association with Cape Nature in the early 2000s. Approximately 63% had visited the Groot Winterhoek Wilderness Area.

3.3.1.7 Associations between landowner values, views and awareness and landowner and land use characteristics

No response variables were associated with harvesting and ecotourism land use. There was, however, a positive association between valuing the Groot Winterhoek Wilderness Area for non-use bequest and option, intrinsic-moral, erosion control and spiritual reasons (i.e. response variables bequest.option, intrinsic.moral, erosion.control and spiritual in Figure 3.8, 1-3 and 8) and personal nature-based recreation (i.e. using privately owned mountain land for personal nature-based recreation, labelled private.recreation in Figure 3.8). Finding merit in the strict protection of the Groot Winterhoek Wilderness Area (GW.merit.protect) was also positively associated with personal nature-based recreation. For example, in Figure 3.8 for results numbered 1-3 and 8, the variable personal nature-based recreation (labelled private.recreation) is green and falls inside the red circle which indicates a positive association with each dependent variable at p<0.05, respectively.

Spiritual-cultural reasons for valuing the Groot Winterhoek Wilderness Area, burning and alien clearing on privately owned land and levels of awareness of local conservation programmes (spiritual, burn, aliens.clear and aware.conservation.progs, 4-7 in Figure 3.8) were all positively associated with the landowner conservancy. Levels of awareness of local conservation programmes were positively associated with years of land ownership (own.years) and whether a landowner had visited the Groot Winterhoek Wilderness Area (GW.visit). Alien clearing was negatively associated with the distance from the Groot Winterhoek (distance.GW.m) and other nearby protected areas (nearest.PA). Burning land for fire management was positively associated with (in addition to the conservancy) living on the mountain (live), years of land ownership, cultivation and grazing (Figure 3.8). There were no associations between the response variables related to mountain and fire policy awareness and influence and all predictors. There were also no associations between valuing the Groot Winterhoek Wilderness for existence non-use value, indirect water-related and direct recreational uses and predictors. Therefore, the results for these response variables are not shown in Figure 3.8.

3.3.1.8 Values, views and land management conditional on land use and landowner characteristics

The odds of valuing the Groot Winterhoek Wilderness Area for intrinsic-moral ($\beta = 1.89 \pm 0.64$, $z = 2.94$, $p = 0.003$), non-use bequest and option ($\beta = 1.49 \pm 0.61$, $z = 2.41$, $p = 0.01$), and erosion control reasons ($\beta = 1.4 \pm 0.64$, $z = 2.18$, $p = 0.02$), were higher for landowners that used their property for personal nature-based recreation (see Table 3.2 for definitions). These landowners were also more likely to find merit in the protection (as opposed to private ownership) of the Groot Winterhoek Wilderness Area than landowners that did not use their properties for private recreation ($\beta = 2.63 \pm 1.12$, $z = 2.34$, $p = 0.01$).

Being a part of the conservancy ($\beta = 4.9 \pm 1.87$, $z = 2.16$, $p = 0.009$), or using land for cultivation ($\beta = 2.793 \pm 1.16$, $z = 2.39$, $p = 0.01$) or grazing ($\beta = 4.12 \pm 1.87$, $z = 2.19$, $p = 0.02$) increased the odds of burning land for fire management and fuel reduction. Furthermore, the odds of clearing alien plants on private land increased for properties closer to the Groot Winterhoek Wilderness Area ($\beta = -0.00007 \pm 0.00002$, $z = -2.63$, $p = 0.008$) or other protected areas ($\beta = -0.0002 \pm 0.00006$, $z = -2.76$, $p = 0.005$). The conservancy ($\beta = 2.86 \pm 1.1$, $z = 2.58$, $p = 0.009$), and number of years of property ownership ($\beta = 0.04 \pm 0.01$, $z = 2.49$, $p = 0.01$) increased the odds of awareness of local conservation programmes. Being a part of the conservancy increased the odds that a landowner would value the Groot Winterhoek Wilderness Area for spiritual-cultural reasons ($\beta = 1.9 \pm 0.73$, $z = 2.7$, $p = 0.006$).

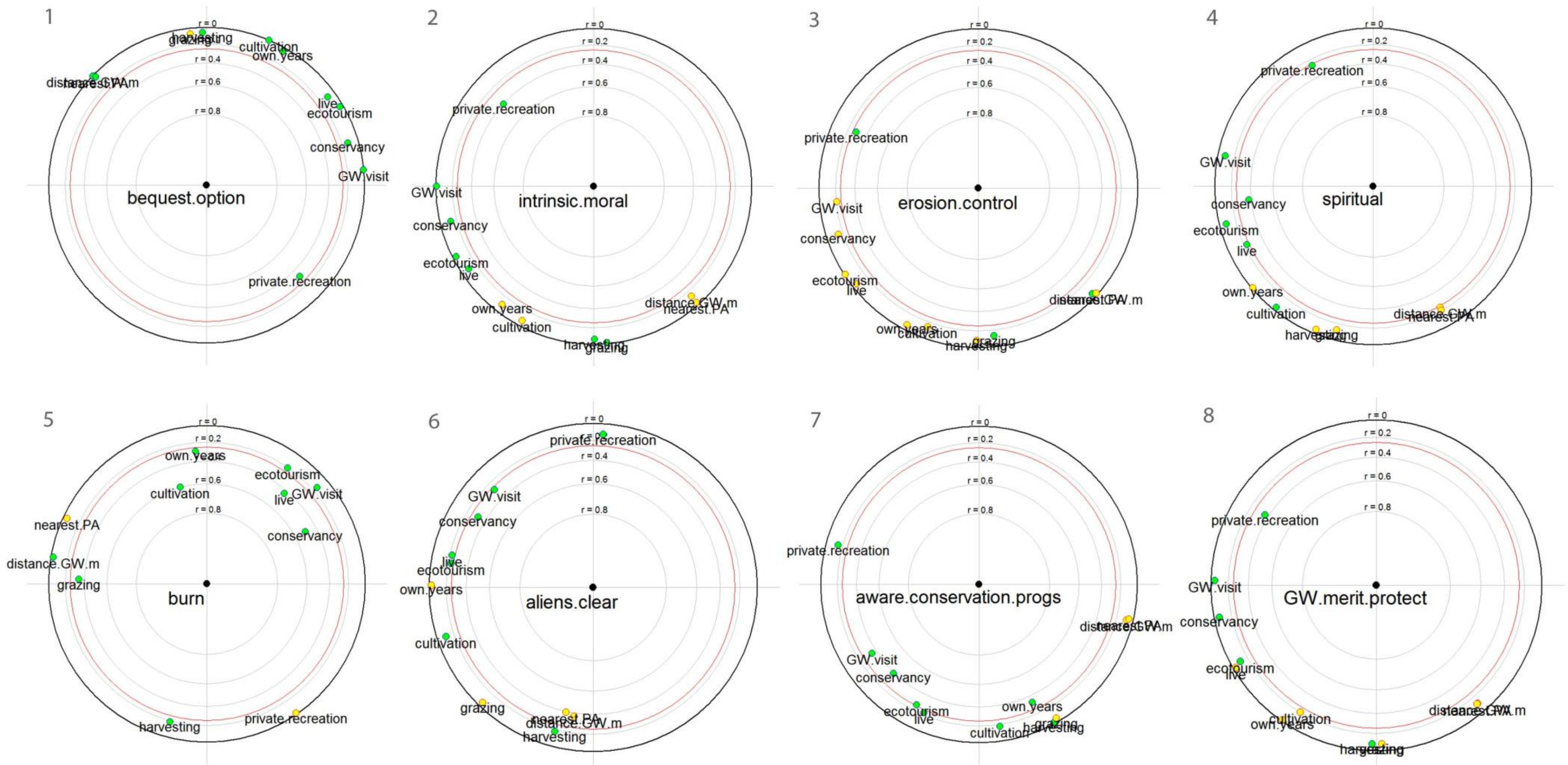


Figure 3.8 Focused principle component analysis for response variables that were associated to one or more predictor variables. Predictor variables inside the red circle are associated with the dependent variable at $p < 0.05$. Green variables are positively associated with the dependent variable and yellow variables are negatively associated with the dependent variable. The following response variables were not associated with predictors at $p < 0.05$ and therefore are not presented: awareness and influence of the National Veld and Forest Fire Act or the Mountain Catchment Areas Act; and existence non-use value, indirect water use value and direct recreational use values.

3.3.2 Qualitative results

3.3.2.1 Past land use (source items = 53, references coded = 63)

Landowners described their mountain property as having been used in the past as grazing for cattle, goats and sheep (35%), for harvesting (37%) or for cultivation (26%). Approximately 32% of landowners referred to burning the veld on their property in the past. The term patch burning was used to refer to burning the vegetation in patches and in certain cases in association with grazing livestock on the mountains. Burning was also associated with the harvesting of buchu (*Agathosma betulina* or *A. crenulata*). Properties situated on the top of the mountain in moderately sloping and flat terrain were largely used for fruit cultivation (11%), in particular cling peaches and apples, from the late 1960s to the late 1980s and into the 1990s. As result of a change in agricultural markets, there was a switch to growing fynbos for cut flowers from the late 1980s and during the 1990s. Recreational activities and tourism were introduced to the area during the late 1990s and early 2000s. A growth in ecotourism and in the recreational use of the mountains is predicted in future. However, this is considered highly dependent on current and future legislation including the implications of the National Veld and Forest Fire Act and other legislation that regulates infrastructure establishment on mountain slopes and especially in river and stream gorges (known locally as 'kloofs') (Figure 3.9 a).

3.3.2.2 Past coverage of aliens (source items = 23, references coded = 25)

Only 38% of the 60 landowners interviewed indicated that they were familiar with alien plants being on their property or near their property in the past. However, dense pine plantations and windbreaks including pines, gums and wattles were described as present in accessible and flat, high-elevational areas during the early 1970s. Although completely private, the pine plantations were described as encouraged and subsidised by government. A substantial reduction in pines in the mountain catchment over the last two decades was described because the concerted efforts to clear alien plants by private landowners. Areas still infested with black wattle (*Acacia mearnsii*) were, however, still noted in the mountain. Pines (*Pinus spp.*) were cited as being easier to clear than wattles (Figure 3.9 b).

Figure 3.9 Word clouds indicating the 30 most frequent words for a) past land and fire use; and b) past alien coverage for privately owned land in the Groot Winterhoek Mountain Catchment. Text search queries were used for each word in the word cloud to analyse the sentences and paragraphs containing these words across all transcripts.

3.3.2.3 Drivers of economic and environmental sustainability in the catchment (Figure 3.10)

3.3.2.3.1 Fire (source items = 36, references coded = 88)

Fire was viewed as a significant driver of change in the catchment. Extensive and damaging fires during the 1990s and 2000s were described in detail. These fires burnt large portions of the catchment and damaged / destroyed the natural vegetation, orchards and recreational infrastructure. Extensive and high intensity fires were associated with increased vegetation cover in the catchment over the last two decades. The implications described included landslips, sedimentation of dams and weirs (affecting water supply from the mountains) and an increase in wild animals on lowland properties. Most fires on the mountain were described as caused by anthropogenic actions on the lowlands with fewer fires being cited as starting from lightning, falling rocks and baboon interference with power lines. Positive aspects of fire were also recognised. This included increased accessibility to the mountain for alien clearing, hiking and other nature-based recreational activities.

The current approach to fire management and control in the mountains was described as reactive as opposed to proactive. This was linked to the current uncontrollable nature of fires. A more integrated approach to fire management between government and private landowners was identified as being an important component to managing fire in the catchment. Controlled burning was seen as a requirement for vegetation management and wild animal protection in the mountain corridor. Back burns were highlighted as efficient firefighting tools. However, fire management activities were considered extremely difficult in mountain terrain and not feasible for private landowners. The financial and legal risks associated with starting a fire on private property and the difficulty in controlling fire in the mountains were identified as problematic. In certain cases, landowners had overcome these difficulties. This included recently conducted controlled burning on their properties with approval from Fire Protection Associations. Certain older landowners still burnt their land regardless of fire policy implications.

Landowners acknowledged that burning should be done in a way that protects the natural vegetation while also ensuring the sustainability of water supplies and protection of wild animals. The impression was that the current large fires were not conducive to the protection required. Information on how and when to burn was identified as being either not consistent, clear or practical in terms of managing fynbos and fire in a mountain context. Developing cells of properties that could then be managed as fire and vegetation units with one fire and vegetation management plan was proposed. Certain landowners felt that specific lessons could be drawn from how older farmers used to burn their land on the mountain in the past in a system known as patch burning.

Many landowners felt that controlled burning should be conducted on the mountains by government. An alternative approach proposed was for private landowners' mountain land to be exempt from certain aspects of the fire policy. Assistance for management plans and firebreaks was highlighted as a requirement. It was recommended that the approach should be built on progress made by Fire Protection Associations, which were seen as integral components to effective management and communication during fire seasons. A smaller number of landowners were completely against controlled burning in the mountain catchment. Cutting the vegetation to reduce the density and then waiting for a natural fire to come through the landscape was preferred. Most landowners, however, were in favour of removing and thinning the vegetation (whether it be through cutting or burning) to reduce unwanted alien and pioneer plants (e.g. *Stoebe* species) and to manage fire risk and fuel loads.

3.3.2.3.2 Money (source items = 28, references coded = 55)

Reduced use of mountain land was linked to increased levels of wealth of landowners bordering the mountain. For example, landowners indicated that there had been an increase in the production of luxury crops such as citrus and wine in lowland areas and referred to mountain land as not being the priority for use in comparison to their lowland portions of land. Past land use in the mountains was described as a source of subsistence income or support. However, increased levels of wealth in nearby cities and globally was cited as an important driver of ecotourism opportunities in the mountains, which had led to an increase in the number of foreign and South African tourists visiting the mountains.

Increased levels of wealth were also cited as the reason for an influx of lifestyle and weekend farmers owning land in the catchment. Lifestyle and weekend farmers comprised both city-based individuals and families that used their properties for personal nature-based recreation during weekends and holidays (a few days to weeks a month) and permanent residents on the mountain that only used their properties for personal nature-based recreation. An increase in lifestyle farmers was, however, also linked to reduced capacity to manage fires and reduced experience in general, in managing mountain landscapes.

Many landowners also referred to the negative financial implications associated with managing and controlling fire in the mountains and especially the hidden costs associated with the National Veld and Forest Fire Act. Financial strain in general was linked to alien plants (in terms of the resources required to clear), fire and market related problems associated with buchu harvesting. Landowners referred to a drop in the price of buchu, which has occurred over the last decade and which has prevented expanding the economic use of mountain land.

3.3.2.3.3 Water (source items = 28, references coded = 37)

Fire and water were closely allied in descriptions of change and effects of change in the mountain. Fire was seen as both a negative and positive influence on water flow from mountain properties. Landowners indicated that a correct balance of vegetation and fire was required to sustain flows and prevent soil erosion in the mountains and other related damages. Limited fire use and the subsequent increase in vegetation cover were cited as reasons for a decline in water from the mountains in recent years. Older landowners referred to burning the mountain in the past and especially burning the gorges to release water for drinking and other purposes. The view that the catchment was being burnt in unplanned and large fires was considered a mechanism of soil erosion and damaging to local and regional water supplies.

Certain landowners referred to the National Water Act (No 36 of 1998) and the complexity surrounding increasing the size of dams or building new dams on mountain properties. A strategy proposed to assist landowners in the management of their mountain land and for protecting water supplies, hinted at a payment for ecosystem services strategy. Specifically, landowners indicated that a portion of the income from the water made in cities and towns should be reinvested in mountain management activities such as assisting landowners manage mountain properties in terms fire, aliens and soil erosion. Landowners also felt that some of this income should also be directed to applied research.

Although landowners acknowledged and described the exacerbating role of climate on fire and water, it was not referred to as the main driving force behind water issues in the catchment. Climate was indicated as a negative contributor to existing challenges associated with water and fire policies and interventions related to vegetation management. For example, specific dry periods and climate events associated with rainfall variability were described. This included the recent drought in 2015 and previous events of limited rainfall or extreme rainfall variability, which were considered to exacerbate the interacting effects of fire and vegetation on water but also enhanced challenges associated with water storage, abstraction and water use efficiencies.

3.3.2.3.4 High intensity agriculture (source items = 22, references coded = 49)

The largest change in terms of cultivation on mountain land has been a shift from fruit to flowers and the introduction of high-income crops such as berries grown at high altitudes. The unique climate on the mountain was highlighted as a rare opportunity for the cultivation of high-income crops that require cold temperatures (chilling units). Fire, however, posed a serious concern for these cultivated crops. Certain landowners have sold their properties or have subdivided their land and sold portions of their properties because of the financial implications associated with fire and the interaction between fire and policy. This has also interacted with an increased demand for mountain properties for lifestyle and recreational focused owners.

The main income source for landowners bordering the mountain and owning portions of mountain slopes is their lowland properties or lowland portions of properties. The mountains, in contrast, are rarely used for income generating activities. For these landowners, intensification and expansion of cultivation in the lowlands along with an introduction of luxury crops on the lowlands is the main reason for the reduction in the use of the mountains for grazing. In some instances, the mountains were considered never to have been used. Certain new or very young landowners could not rationalise the thought of using mountain areas for economic or subsistence activities other than recreational based tourism.

A major inhibitor to the intensification or the expansion of agriculture on the mountain is the poor quality of roads as well as the nature of the terrain and low productivity of the area. Although the main access road was upgraded and tarred recently in the last decade there is still a large section, which is gravel. The road has been and is still a major hindrance in terms of building the agricultural economic value of the mountains. Certain landowners are of the impression that the Groot Winterhoek Mountains have the potential to look like the nearby Piketberg Mountains in terms of agriculture if the access route was improved. For certain property owners (especially lifestyle and weekend farmers) the limited access is viewed in a positive light as it limits the ability to develop the mountains. This is seen as a mechanism that protects the mountain as a corridor for wildlife movement. The current access via the road is not a critical factor for lowland landowners bordering the mountain who own portions of the slopes. However, many indicated that while they would like to access portions of their land this was not possible for financial and legal reasons.

3.3.2.3.5 Recreation and tourism (source items = 21, references coded = 42)

Landowners noted that there has been an increase in lifestyle farmers as well as ecotourism and nature-based recreation in the catchment. Increases in wealth and living standards in the cities and lowland areas as well as a greater appreciation for nature-based activities by a new generation of landowners in the catchment were seen as major influences of this change. This included an increase in awareness of the importance of conservation and an increase in the appreciation of nature-based recreational activities. It was suggested that the older generation of landowners used their mountain properties for marginal economic income and subsistence purposes. The goal was to survive and there was no time for recreation or appreciation of nature-based recreational activities in the past.

Tourism, ecotourism and agritourism were cited as potential desired future economic activities for many of the current landowners. Despite the indication that ecotourism and recreational based activities in the mountains would increase in future, existing fire and environmental legislation was described as a major inhibitor of growth in these activities. The potential for using the mountains for tourism and recreational based economic and private activities was described as dependent on the interactions between the effects of fire in the landscape and the current as well as future fire, environmental and land policies. Many landowners queried the controls and processes currently in place to build structures and access roads in the mountains.

3.3.2.3.6 Conservation (source items = 20, references coded = 53)

A portion of the landowners interviewed described a landowner conservancy that was set up in the early 2000s and linked this to an increase in the appreciation and awareness of conservation activities over the last decade and a half. The conservancy comprised a formal, voluntary group of landowners in the mountain catchment that was established through communications with Cape Nature. Landowners noted that it had improved the way in which private landowners communicated and collaborated around economic activity, agriculture infrastructure, alien clearing, access roads, controlled burning, fire management and conservation. A few landowners did not share these views and expressed that the large agricultural focus of the conservancy was not beneficial for conservation. When referring to the conservancy, however, many landowners did so in a positive sense. Many were also of the view that aspects of conservation in the mountains could be enhanced by more strategic work conducted by Cape Nature.

3.3.2.3.7 Aliens (source items = 18, references coded = 30)

Many landowners referred to the activities undertaken in the area by Working for Water. Some landowners viewed the programme as exceptional. For others, however, the alien plant clearing teams were not viewed as being effective in river and mountain catchment areas that require constant follow-up clearing and attention to detail. Teams were seen as being poorly paid and as a result, at times not motivated to clear effectively in the dangerous and difficult terrain.

Landowners, particularly within the mid-sections of the catchment and forming part of the landowner conservancy, felt that advances had been made in the clearing of alien plants since the 1990s. Reference was made to the removal of pine plantations and the exotic windbreaks which were planted on the mountains in the past and the resultant invasions. Initial clearings were attributed largely to the efforts of private landowners. Landowners with extremely rugged terrain or bordering the mountainous area and owning mountain slopes (especially in the southern section of the catchment) felt that aliens were getting worse every year and that it was impossible as a landowner to manage the problem. This included properties which had large river sections with deep gorges, and which were cited as being infested with wattle species. For the far northern sections of the catchment, landowners felt that aliens were not a problem and did not see it as a major threat to the area.

Controlled burning was seen as a potential mechanism to assist in the clearing of alien plants from mountain slopes, as it would increase accessibility and stimulate the germination of seed banks. However, cutting alien plants and burning them in stacks was preferred over controlled burning by certain landowners and in particular by lifestyle and weekend owners. Many landowners placed pioneer indigenous plants, such as *Stoebe* spp. and *Cliffortia* spp., in the same category as alien plants and felt that they should be regularly cleared. Some landowners believed assistance for either burning or alien plant clearing should be provided to private landowners with mountain land.

3.3.2.3.8 Government (source items = 16, references coded = 44)

Past ‘champions’ from local and national government departments were mentioned often during the interviews with landowners. This was specifically in relation to managing and controlling fires and conducting controlled burns, back burns and firebreaks. Certain older landowners indicated that current fire and predator control policies were developed without integrating local landowner views. However, most landowners referred to the Fire Protection Association in a positive manner, highlighting the private nature of the funding arrangement and its efficient organisation. The shift of the financial burden of runaway fires in the mountains from government agencies to private landowners was viewed as negative for the management of vegetation and fire in the catchment. In addition, recent landowners were less aware of the role of government in the past and often had relatively limited views on current policies.

The government agency most suitable to assist private landowners was identified to be at a regional or even local level with many landowners referring to Cape Nature. Certain landowners were extremely positive of their engagements with Cape Nature while others were less so. Negative perceptions related to the lack of management expertise and depth of knowledge within Cape Nature managers although the current constraints placed on a conservation agency when competing against funding requirements of other government departments was recognised. It was further acknowledged that national government departments have greater immediate concerns such as addressing issues relating to poverty, crime, housing and health, than fire and alien management in water catchments. Funding issues were predicted to be a continued problem in future.

3.3.2.3.9 Market (source items = 14, references coded = 23)

Local and international markets were described as having an important influence on land use practices in the catchment especially in terms of wildflower harvesting and buchu harvesting. Approximately 30% of landowners interviewed indicated that they had stopped harvesting buchu from their mountain properties over the last ten years because of a drop in the market price. The buchu price was cited as being exceptionally high (e.g. ~R40/kg) during the 1990s and early 2000s. This is in comparison to the current market price (~R10.00/kg). Several interacting global and local factors were noted as being responsible for fluctuations in the buchu market (see 3.3.2.3.14 Harvesting below).

Older landowners noted that in the past the mountain land was used for subsistence food for family and workers or for subsistence income. This was no longer required. The reduced demand for wool was also mentioned as a large influence in the reduction of sheep on the mountains. Increases in wealth linked to growing high-income crops (such as citrus and wine) on lowland properties and the intensification of agriculture based on local and international market demands were also referred to as a reason for the limited use of the mountains for pasture and other economic uses. The general perception from the landowners was that there is no longer a need to supplement food or income sources with marginal farming activities in difficult and relatively unproductive mountain areas.

3.3.2.3.10 Awareness (source items = 13, references coded = 23)

Conservancies that bring together landowners to discuss conservation and land use issues in the mountain were seen as important mechanisms for increasing awareness, communication and management effectiveness in relation to fire, aliens, water and conservation. Landowners highlighted that the new generation of landowners are more predisposed and accepting of conservation ideas and therefore, more receptive to the formation of conservancies. Enhanced integration with other existing agriculture committees and water boards was cited as an important next step that would require careful consideration. There were certain levels of informal integration that already existed, however; no formal process was in place to support collaboration between different entities in the landscape. Although the role of Cape Nature in creating awareness and collaboration was identified as important and effective by some landowners, not all landowners shared this view. Certain landowners were of the impression that Cape Nature was not visible in the landscape due to funding and human capacity constraints. Others shared negative views related to Cape Nature's technical capacity to provide guidance on fire and vegetation management. In addition, certain landowners felt that Cape Nature was not interested in developing collaborations with private landowners in the area.

Certain older landowners indicated that in the past, mountain areas were managed relatively well in terms of communication and collaboration between farmers. Several interacting influences were seen to have reduced flexibility and communication between landowners. The current fire policy was cited as one example which had at times increased tensions between neighbouring landowners and resulted in numerous and complex legal situations. For example, certain landowners described multiple interacting legal cases that had occurred whereby landowners affected by fire attempted to recoup their financial losses by proving that their direct neighbours had been negligent in terms of the National Veld and Forest Fire Act. However, fire pathways can be complex and can cross many properties. As a result, more than one legal case associated with many different landowners had occurred in the past. The potential for future legal implications and costs associated with fire occurrence and negligence in terms of the National Veld and Forest Fire Act has contributed to heightened tensions between landowners in the catchment.

3.3.2.3.11 Wildlife (source items = 13, references coded = 21)

Landowners referred to wildlife in the mountains in both a positive and negative manner. Baboons and leopards were mentioned frequently, and landowners connected movement of wildlife and issues associated with wildlife to fire and vegetation cover. Landowners indicated that the large fires sweeping through the catchment because of the large fuel loads had been detrimental to the wildlife on the mountains primarily because of the lack of shelter and shade available for wildlife during and after extensive fire events. The increase in vegetation and thickness of the vegetation on the mountains was viewed as a problem for the movement of smaller wildlife e.g. klipspringer, grey rhebok and hyrax. Many landowners indicated that after large fires livestock mortalities due to leopard increased and that there were more sightings of leopards in the lowlands.

Landowners that held a negative view of the presence of leopards in the mountains mentioned that in the past landowners were provided far more scope to implement control measures for damage causing wildlife (e.g. caracal, baboons, black-back jackals and leopards). Recent limitations, which were imposed since the early 1990s, were cited as a reason for the decline in use of the mountains for grazing. In addition, this was viewed as the reason why leopards and other damage causing wildlife had proliferated in the catchment with negative effects for smaller wildlife species. In contrast, landowners that viewed leopards as a positive aspect to the area highlighted the importance that the mountain catchment plays as a corridor for protecting leopard and other wildlife movement patterns.

3.3.2.3.12 Vegetation (source items = 13, references coded = 20)

An increase in the density of the vegetation on the mountain was a reoccurring theme in interviews. Landowners mentioned that the mountain vegetation has been left to become extremely thick which poses a fire hazard and has resulted in high intensity and uncontrollable fires in the past. This was translated into negative effects associated with rainfall and the amount of water flowing off the mountains. Landowners referred to local flooding, erosion, and landslips that affected their lowland properties after the occurrence of high intensity fires in the mountain areas.

The occurrence of indigenous *Stoebe* species (referred to generically as slangbos) was mentioned frequently in a negative light, with many landowners equating slangbos to an alien species. Landowners indicated that in certain areas of the catchment, *Stoebe* had become incredibly dense and that to restore the "natural" fynbos the plant had to be cut out and then burnt in piles. Lifestyle owners or owners that use their land for recreation (both private and ecotourism) indicated that the proliferation of *Stoebe* and other pioneer plants was negative for recreational activities. Landowners also indicated that it was extremely flammable and presented a fire risk. Another problem plant mentioned frequently was *Cliffortia* spp.

Certain older landowners mentioned that in the past the mountains were far more open and free of thick vegetation. This resulted from the ignition, in the past, of frequent smaller fires that created a patch mosaic of vegetation types on the mountain. Landowners indicated that there were also numerous cattle and goat paths in the mountain in the past, but these had now completely disappeared. The thickness of the vegetation was also linked to a proliferation of ticks in the recent decades, which was a nuisance for livestock and for their recreational activities.

3.3.2.3.13 Lowlands (source items = 11, references coded = 13)

The lowlands were mentioned frequently by landowners that lived on the border of the mountains and owned both mountain and lowland properties. In the past, certain landowners used their mountain properties for grazing and for the harvesting of wildflowers. However, an increase in high income and luxury crops in the lowlands was linked to the reduced need to use the mountain slopes for subsistence as well as marginal economic activity. Landowners noted that these areas were not suitable for agricultural activities and furthermore required substantial effort in relation to their economic returns in comparison to the lowlands. Almost all landowners viewed their mountain properties as completely environmentally sustainable with many indicating that economic returns are rarely a goal for the property or irrelevant because of steep slopes making the land non-viable for agricultural economic activity. The link between fires coming from the lowlands was also highlighted by landowners. Lowland fires were seen as being caused because of either negligence or ignorance.

3.3.2.3.14 Harvesting (source items = 9, references coded = 13)

The influence of the market price on the harvesting of wild plants was a recurring theme for buchu harvesting. Fire was also identified as an important driver of change in harvesting. Landowners indicated that when the price for buchu was high, the market demand was only for one species, *Agathosma betulina,* due to its high-quality phenols and oils. However, in the study area *A. betulina* is not as common as *Agathosma crenulata, which* also occurs on the mountain. Certain landowners attempted to increase profits by mixing *A. crenulata* with *A. betulina* harvests. At the same time, there was a substantial increase in poaching and illegal and unsustainable harvesting practices due to high market prices. This further reduced the quality of buchu products. Unsustainable harvesting practices by landowners interacted with poaching levels during this time. Landowners were of the view that the target plants would be harvested regardless whether it was by owners of the land or illegally by poachers. The planting of buchu on the lowlands also increased and this was considered to have further reduced the price of buchu because of the increased availability of buchu products. Subsequently there has been a significant drop in the buchu market price driven by international European markets. This occurred because the strong demand for high quality buchu products from European countries, especially Germany, created an initial spike in buchu prices. The phenols were highly sought after for perfumes as well as black current cool drinks. However, when the buchu imports arrived in Europe the poor quality was detected and buchu imports were subsequently boycotted. This resulted in a decline in the market price of buchu. Many landowners have completely stopped or reduced their harvesting levels, as it is no longer financially viable. The poaching of wild buchu material, however, has continued.

3.3.2.3.15 Grazing (source items = 6, references coded = 11)

Certain landowners referred to using the mountain as a safety net for their livestock during times of drought. For example, landowners particularly relied on certain portions of their mountain land for grazing their livestock during the drought in 2000 and again in 2015 and 2016. This is because there was not enough feed on the lowlands and therefore they moved their cattle into the mountains to feed. Landowners that were familiar with using the land for grazing in the past indicated that to use the vegetation on the mountain for grazing it needed to be burnt. Landowners noted that in the past the vegetation on the mountains was more open and easier to walk through because of the frequent burning and due to the increased number of livestock paths and especially goat paths. Goats were described as extremely efficient at utilising mountain fynbos vegetation and therefore were the dominant livestock pastured on the mountain in the past. The decrease in livestock grazing, and in particular goats, was associated with the reduced demand for marginal income and subsistence food for families and workers. This was related to increased living standards and agricultural efficiencies on lowland properties. An increase in damage causing wildlife policies was viewed as an additional reason for the reduction in grazing on the mountains. The regulation of measures relating to the control of damage causing animals such as leopards were cited as creating greater financial risks associated with grazing livestock on the mountains.

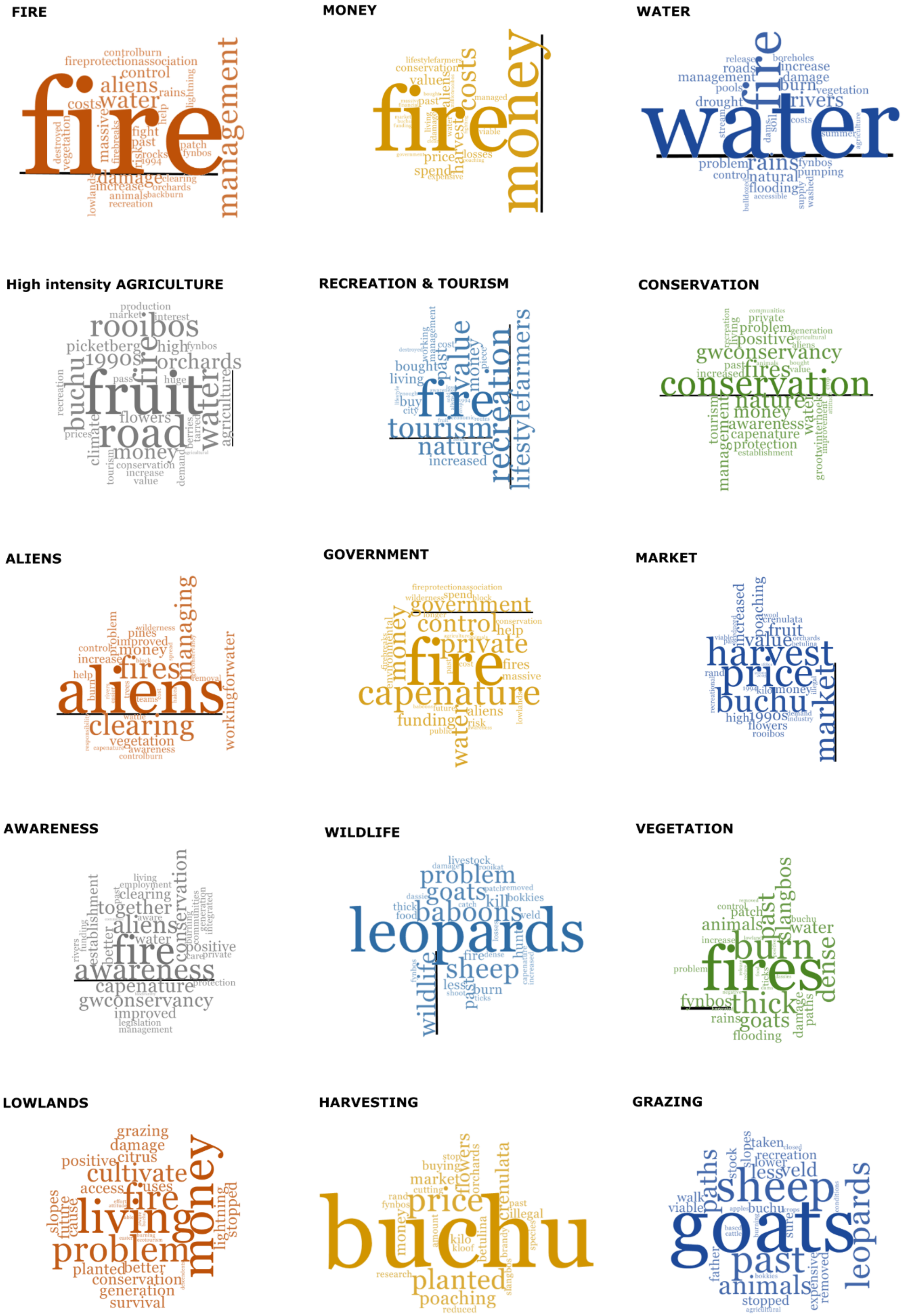


Figure 3.10 Word clouds showing the 30 most frequent words for each theme coded for descriptions of change drivers of change as perceived by private landowners in the mountain catchment. Text search queries were used for each word in the word cloud per theme to analyse the sentences and paragraphs containing these words across all transcripts.

3.3.2.4 A conceptual model of drivers of land use change and land use transition pathways

From the interviews with landowners and drawing on the concept of land use transitions and associated explanatory frameworks (Lambin et al. 2001; DeFries et al. 2004; Foley et al. 2005; Lambin & Meyfroidt 2010) a conceptual model of the local socio-economic and socio-ecological drivers of land use change for the mountain socio-ecological system was developed (Figure 3.11) The model summarises i) how land use has changed over the last forty years; ii) the local-level socio-economic and socio-ecological drivers of land use change; iii) the consequences of land use change including the ecological responses and related feedbacks; and iv) the fundamental forces that have influenced local-level land use decisions.

The model is comprised of four main sections. The first section considers the dynamics around exogenous socio-economic drivers which have influenced land use change in the area. It identifies five components which have influenced land use change in the area. These are: i) increased living standards and wealth; ii) attention to environmental factors including concern for protecting the environment (referred to as "environmentalism"); iii) technology innovation and transfer; iv) market fluctuations (including growth and failure); and v) national and regional policy. The second section comprises local-level socio-economic and socio-ecological effects in relation to exogenous socio-economic dynamics. This section condenses the narratives concerned with the drivers of change as perceived by landowners. It also creates links to the third component of the model which summarises the quantitative results of land use change. The fourth component of the model summarises current ecological responses and potential feedbacks resulting from land use change. One endogenous socio-ecological feedback is identified in the model as regulating land use change in the catchment over the last forty years. This relates to the intrinsic ecological constraints inherent in mountainous areas which are not viewed as prime agricultural land. This is captured in the model as a separate aspect and is not included under the four components described above.

Three land use transition pathways are identified in the model as defining the way in which land use, local drivers and exogenous socio-economic dynamics have changed over time. These are: i) economic growth and development (including changes in the market economy); ii) globalisation; and iii) natural resource scarcity and are described in more detail below.

3.3.2.4.1 Economic growth and development

The results of this study suggest that agricultural intensification resulted from an increase in market demand. This was concentrated in the most suitable areas such as the lowland portions of properties and certain high lying flat areas in the mountains where luxury crops such as wine, citrus and berries could be cultivated. This process required an adjustment of agriculture activities to land quality, which in turn, interacted with the endogenous socio-ecological conditions as well as the intrinsic ecological constraints of mountain land. Farmers adopted productive and efficient agricultural technologies, which increased crop production in the lowlands and resulted in the introduction of luxury crops to the area. This was likely spurred on by additional influences associated with increased water consumption in cities and the perceived competition for water between the urban and agricultural sector. This not only influenced dam building but also the implementation of improved irrigation efficiencies for economic reasons. Grazing the mountains was no longer required by landowners who owned land on the lowlands, as it was not profitable in comparison with their lowland activities. Over time, there has been an increase in wealth and living standards of landowners in the study area largely because of an increase in income. This increased wealth has further reduced grazing activities in the mountains and increased the appreciation of personal nature-based recreation and ecotourism levels through an interaction with the growth of environmentalism.

3.3.2.4.2 Globalisation

As South Africa's economy has become increasingly integrated into global markets, there has been an increase in commodities, labour, capital, tourism, technology and ideas. Local economic development and growth associated with becoming increasingly globalised has further stimulated local innovation and technologies. New technologies and international markets directly influenced agricultural intensification in the study area and incentivised the production of luxury crops. It has also lead to the shift of agricultural practices to areas of prime agricultural land while grazing was abandoned in the mountains due to intrinsic ecological and topographical constraints. Environmentalism and conservation ideologies, which are strongly influenced by global trends and developments, were transferred to local levels and to new generations of landowners. Nature-based tourism was increasingly recognised as a worthwhile industry and offered to foreigners as well as South African based tourists. National policies were further influenced by the manifestation of conservation ideologies in the public and within regional departments especially in relation to damage causing wildlife control policies.

3.3.2.4.3 Natural resource scarcity

National and regional policies, concerned with the conservation of mountain catchments from the 1940s to the 1980s, were formulated and implemented as a response to the perceived adverse impacts of specific land use types. These included the burning and grazing of mountain catchments which were considered important for the water supply in South Africa and specifically in the Cape Floristic Region. Government officials perceived the effect of these burning and grazing activities to be a reduction in suitable vegetation cover and a subsequent increase in soil erosion with negative impacts on water supplies. Government officials and researchers were also likely influenced by environmentalism trends especially related to soil erosion and desertification, which were prevalent globally from the early 1900s to the 1960s. Despite efforts by successive administrations, the influence of policies since the 1970s on the reduction of the use of mountain land for grazing has been overshadowed by globalisation and associated local economic development and growth. This has resulted in an intensification of agriculture and the production of high-value luxury crops in the lowlands primarily for economic purposes. The process of globalisation and local economic development and growth likely interacted with increased competition for water in the lowlands. It likely occurred because of increased demand and consumption in nearby cities and towns as well as an increase in rainfall variability.

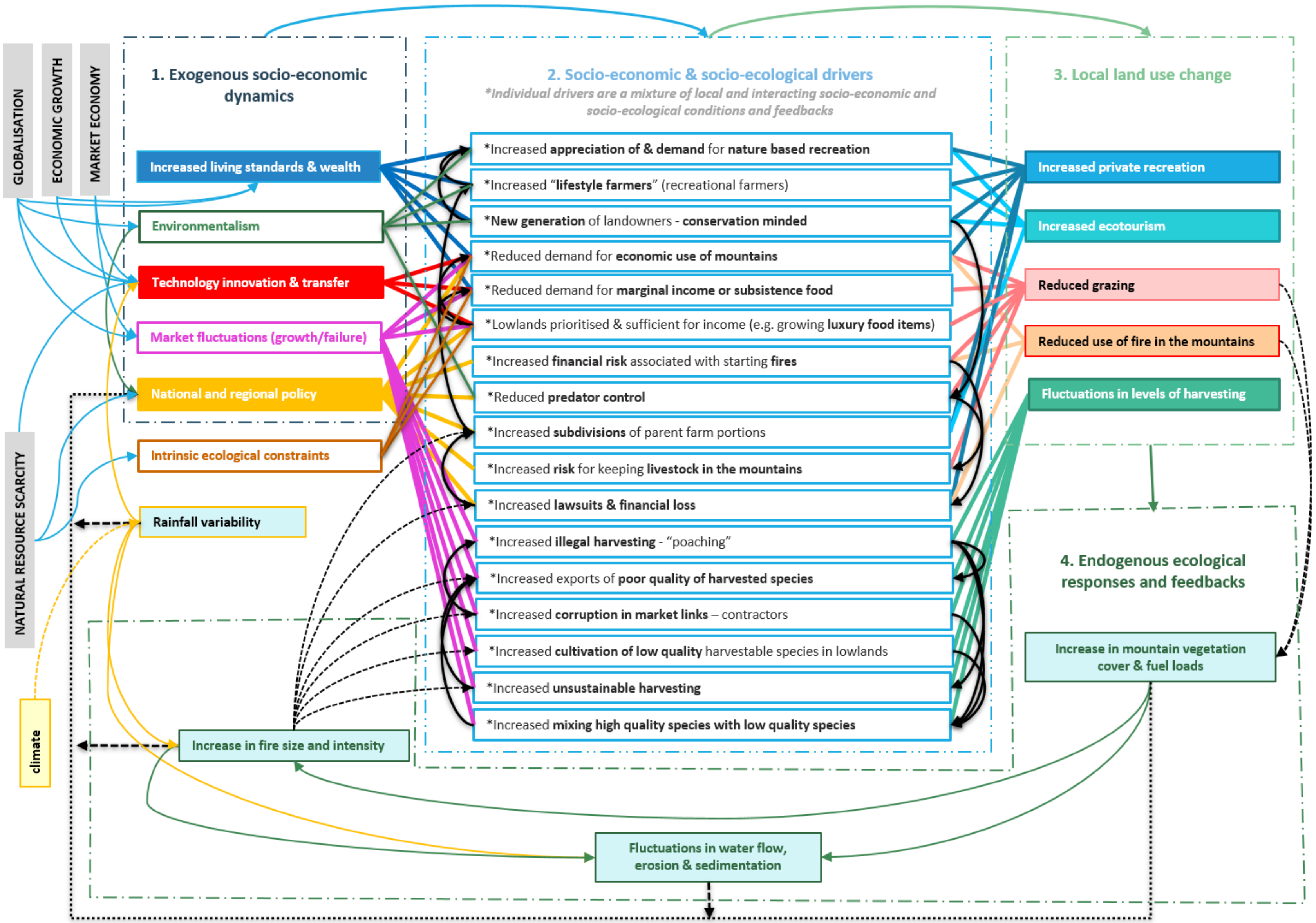


Figure 3.11 Private land use transitions in a mountain catchment in the Cape Floristic Region over the last four decades including external and internal processes that have influenced land use transitions and ecosystem responses within the mountain social-ecological system. The diagram shows exogenous socio-economic dynamics, local socio-economic and socio-ecological effects and ecological responses as well as associated interactions and feedbacks (interpreted using perceptions of current private landowners integrated with repeat quantitative measures of land use change). Interactions can be one- or two-way, depicted by arrows versus straight lines respectively, but depend on the circumstances and the resulting land use transitions and ecosystem services trade-offs. This conceptual diagram focuses on change

3.4 Discussion

3.4.1 Land use transitions, causal mechanisms and ecosystem service trade-offs

Human societies have followed a sequence of different land use regimes described as starting from natural ecosystems or pre-settlement wildlands progressing to frontier clearing, followed by subsistence farming practices and then lastly to intensive practices of agriculture accompanied by urban areas and protected biodiversity or recreational focused areas (Foley et al. 2005). The mountain case study presented here is an example of a land use transition from subsistence marginal farming to an increase in the importance of mountain land for cultural ecosystem services, especially personal nature-based recreation and ecotourism. The transition has also included an intensification of agricultural practices in suitable more productive areas and the abandonment of mountain land for grazing. Globalisation and economic growth and development has had a significant influence on the socio-economic dynamics of the region. These were identified as the main causal mechanisms of land use change over the last 40 years and supports substantial case study evidence on land use transitions, which show that people's responses to economic opportunities as influenced by global factors drive land use change (Lambin & Meyfroidt 2010).

The marginalisation of mountain land over the last 50 years, through similar pathways of change as identified in this study, has been recorded in European and other mountainous regions. The intensification of agriculture, as well as increased productivity in flat or valley areas due to improvements in technology, together with the inability to modernize mountain land, due to land steepness or poor access, has been indicated in many instances (MacDonald et al. 2000; Parés-Ramos et al. 2008; Keenleyside et al. 2010; Lasanta et al. 2017, 2017). In the case study presented here, such influences included the exchange of and exposure to new commodities, markets, ideas, technologies and conservation ideologies that resulted in complex upland-lowland transitions and related socio-ecological feedbacks. Similarly to European case studies, local socio-ecological drivers such as the inherent ecological constraints also played a significant role in land use change (Gellrich & Zimmermann 2007; Müller & Kuemmerle 2009; Corbelle-Rico et al. 2012). Local ecological constraints such as rockiness, steepness, and poor soils interacted with exogenous socio-economic dynamics to reinforce the abandonment of grazing activities in the mountains and the intensification of agriculture in the lowlands and suitable mountain sites.

Although the global diffusion of environmental attitudes and values ("environmentalism") has been linked to sustainable land use transitions in many regions of the world (Lambin & Meyfroidt 2010), the influence of environmentalism on trade-offs in land use and associated provisioning and cultural ecosystem services is not well described in the farmland abandonment literature (Lasanta et al. 2017). Place attachment as a factor of mountain farming permanence has been investigated (Hinojosa et al. 2016). However, the growth of personal nature-based recreation as a land use and the influence of this on views and values of conservation has rarely been considered. A shift from domestic livestock farming to wildlife-based ecotourism since the 1980s has been recorded for the drier, non-mountainous parts of South Africa and Namibia in response to economic markets and growing preferences for cultural over provisioning ecosystem services (Smith & Wilson 2002; Scholes & Biggs 2004). A key difference for this case study and a theme which is likely transferable to other similar mountainous and non-mountainous regions, is that the preference for nature-based recreational activities is not centred on wildlife but rather on hiking, and the appreciation of the natural flora of the region.

The diversity of local and global interactions between causal mechanisms of land use change has been recently highlighted for European mountain regions (Lasanta et al. 2017). In contrast with studies that have shown local anthropogenic land degradation as a mechanism of farmland abandonment (Lasanta et al. 2006a; Nadal et al. 2009; Nadal-Romero et al. 2013), this case study emphasises the importance of exogenous socio-economic drivers as opposed to local drivers associated with land degradation. Historical concerns about the declining condition and increased scarcity of natural resources in the mountains of the Cape Floristic Region likely triggered a policy response during the late 1960s and 1970s (Wicht 1943; Ross & Tempel 1961, 1961; Pooley 2012, 2015; Bennett & Kruger 2015; van Wilgen et al. 2016a). The direct link between national and regional mountain and conservation policies, prior to the 1980s, and change in land use, over the last forty years, however, is complex and was not immediately apparent in this case study. It is more likely that the uptake of new technologies, agricultural intensification and increased agricultural efficiencies were due to increased exposure to international markets, local economic growth and development and possibly indirectly linked to competition for water on the lowlands.

There has been a substantial growth in water demand and consumption since the 1970s in nearby cities and towns and within the farming landscape itself. This likely interacted with climate variability, globalisation, economic growth and self-interest to encourage private landowners to intensify their agricultural production practices (Callaway et al. 2008). In addition, prior to the 1990s, South Africa had several different programmes in place to manage predation from damage causing wild animals (so-called “problem animals”) through lethal and nonlethal management. This was carried out in close cooperation with livestock farmers and included the payment of bounties and introduction of other incentives for controlling predators. Such programmes have largely fallen away and have been replaced by a shift to nonlethal management and the vilification of so-called hunting clubs that were actively engaged in controlling problem animals in the past (Bergman et al. 2013). Policies developed in the mid-1990s to control damage causing animals affected a change in predator management activities and, as a result, reduced the viability of using mountain land for grazing. The absence of regular visits to the mountains for grazing purposes, in turn, has also been linked to the reduced use of fire in the landscape.

Despite not being identified as a major causal mechanism of land use change, socio-ecological feedbacks linked to natural resource degradation were still important for influencing specific interactions and affecting land use change. This is particularly the case for wild plant harvesting whereby global and local market forces interacted with ecological degradation and natural resource scarcity to bring about an initial growth in the market but also led to its subsequent decline. These dynamics have resulted in fluctuations in harvesting levels in the catchment over the last forty years. The wildflower harvesting industry in other areas of the Cape Floristic Region has faced similar pressures, including conflicting issues between preserving the value of the region and the harvestable species while also protecting local livelihoods and economic interests (Blokker et al. 2015; van Wilgen et al. 2016a).

At present, projections of farmland abandonment are limited by available information. For example, models are overly deterministic in that they do not consider social and cultural factors that may cause a landowner to continue using their land for semi-economic or uneconomic reasons (Keenleyside et al. 2010; Lasanta et al. 2017). For example, a shift to using mountain land for private recreational activities is not well described in existing farmland abandonment studies and is currently not captured in existing models. Thus, mountainous or remote areas with marginal agricultural potential or production are consistently identified as areas most likely to be abandoned (Keenleyside et al. 2010; Price et al. 2015). This study reinforces the importance of considering many different scales of drivers and effects when investigating the process of farmland abandonment and for predicting future scenarios (Lasanta et al. 2017). The research also highlights the role of cultural ecosystem services such as personal nature-based recreation and ecotourism in mountain socio-ecological system configurations.

3.4.2 Views and values of conservation

There is no doubt that direct nature-based tourism and recreation in protected areas provide key sources of revenue and societal support for protected areas thereby increasing their socio-ecological system resilience (Dharmaratne et al. 2000; Chan et al. 2012a; Daniel et al. 2012; Maciejewski et al. 2014). However, non-use economic and intrinsic-moral values could be just as, or even more, important for connecting people and nature and for building support for conservation and protected areas (Chan et al. 2012b; Abson et al. 2016). It is clear from this analysis that people place value on the protection of a specific area for non-use and intrinsic-moral reasons regardless of the direct or indirect use value of cultural ecosystem services (Chan et al. 2012b). In the context of this case study, it is likely that landowners did not express direct recreational use values for the protected area nearby as they had direct access to personal nature-based recreation. However, direct access to personal nature-based recreation likely played a role in stimulating non-use and intrinsic-moral values for conservation (Theodori et al. 1998; Schultz et al. 2005; Abson et al. 2016).

Mixed results exist in the literature on the relationship between nature-based recreation and conservation values, views and behaviour (Theodori et al. 1998; Farmer et al. 2016). Some studies identified recreational benefits as important drivers of conservation behaviour (Theodori et al. 1998; Koontz 2001; Brenner et al. 2013; Farmer et al. 2016) while others found such benefits to be negligible (Bourke & Luloff 1994). There is similar disagreement in terms of the relationship between environmental values and conservation behaviour. However, certain studies have shown that pro-environmental behaviour can be influenced by environmental attitudes, beliefs, and values (Schultz et al. 2005). This study showed associations between personal nature-based recreation and the valuing of formal protection for non-use (bequest and option) and intrinsic-moral reasons as well as indirect economic use values such as soil erosion control. Furthermore, landowners who used their land for personal nature-based recreation were more likely to find merit in the formal protection of mountain land.

Not all individuals are in the position to access natural areas for recreational based activities. In addition, living standards or wealth can play a role in an individual's view on whether to value the use of natural areas for private recreation or conservation activities (Koontz 2001). Nature-based tourism and recreation can be achieved in both formally protected areas, which strive for conservation of biodiversity, as well as in semi-formally protected areas, which strive for the development and conservation of biodiversity in novel and hybrid ecosystems (Hobbs et al. 2014; Abson et al. 2016). The link between different types of nature-based recreational activities, and views on conservation and values of formal protection in the Cape Floristic Region should be explored further using non-monetary and qualitative approaches.

3.4.3 Conservation in mountainous regions

Despite being tightly linked to processes on the lowlands, and thereby interacting with global and regional socio-economic drivers of change, mountain landscapes are unique landscapes and require different approaches to management and protection (MacDonald et al. 2000). This reiterates the thinking of the South African Department of Forestry in the past when they initially conceptualised the Mountain Catchment Areas Act (Ross & Tempel 1961). However, there has been considerable confusion over the last two decades surrounding the administration and management of land legislated under the Mountain Catchment Areas Act including links with other relevant acts such as the National Veld and Forest Fire Act of 1998 (Rabie & Burgers 1997; Bennett & Kruger 2015; van Wilgen et al. 2016a). The lack of administration and management of the Cape Floristic Region's mountainous upland environments and limited awareness and influence by government in this area is clear from this study. While conservationists in the Cape Floristic Region are currently striving to incentivise conservation on private lands, they are not engaging with existing mechanisms such as the Mountain Catchment Areas Act in the process.

Being an active member of an environmental organisation has been shown in the literature to be an important predictor of conservation interest on privately owned land (Brenner et al. 2013; Farmer et al. 2016). This study supports this finding and highlights the importance of landowner conservancies for conservation efforts. Membership of the landowner conservancy was a predictor of awareness of conservation programmes. It was also perceived to be a positive aspect in the landscape for communication and conservation. For this case study, landowner conservancies can be viewed as an important mechanism through which controlled burning practices and future fire and mountain policy revisions could be further explored and formalised. The importance of actively communicating with landowners the policy and way forward for managing mountain vegetation is an imperative to manage unforeseen negative socio-ecological interactions whether the ecological response is perceived or real. There seems to be a need to move from the current reactive approach to fire in the mountains to a more proactive approach. Although the development of Fire Protection Associations is considered a major advancement for fire management and control and especially for lowland properties, when it comes to mountain properties, there is a definite need, as expressed by landowners, for greater input and a slightly revised approach to fire management and control.

Scenario planning could play a key role in determining future land use transitions and ecosystem services trade-offs given specific plausible climate pathways and interactions with exogenous drivers such as globalisation and economic growth (Peterson et al. 2003; Gillson & Marchant 2014; Price et al. 2015). For example, mountainous regions that provide the necessary chilling units for the growing of some deciduous fruits could be sought after for agricultural practices under future climate conditions. In the context of this case study, technology transfer and innovation could in future open avenues for using the mountains more intensively. This is especially related to the unique climate the mountain provides for luxury food items such as fruits and berries particularly under future climate projections of decreasing rainfall and increasing temperatures for the area. Future scenarios of land use change could also include growth in ecotourism and personal nature-based recreation. On the other hand, there could be a downward swing in ecotourism and private recreation because of lack of access and increased fire risk related to the current fire policies in the landscape. Controlled burning of the vegetation to protect high altitude crops and cut flowers and other luxury food items as well as lowland cultivation and settlement activities could also increase in future. The use of controlled burning could also be influenced by the potential effects of climate change on post fire vegetation recovery rates. Depending on the context of these controlled burns, there could also be potential negative effects for biodiversity and ecosystem requirements.

3.5 Conclusion

The findings of this section emphasise the role that socio-economic pathways such as globalisation and economic growth (including environmentalism) have had on land use transitions over the last forty years as opposed to the influence of natural resource limitations associated with local degradation. The Mountain Catchment Areas Act, while relevant on paper, is not widely recognised by private landowners that own the land. A vast portion of the private land in this case study is used for private recreation or is not used at all due to the rugged nature of the catchment. Given the importance of mountain regions for water supply as well as plant and animal movement corridors, and the role they can potentially play under future climate conditions, the Mountain Catchment Areas Act should be reviewed and potentially revised, updated or replaced. This study highlights the importance of considering probabilistic socio-economic factors that encourage the emergence and growth of personal nature-based recreation and ecotourism in land use change studies associated with farmland abandonment in mountainous regions.

Section 4. The impact of a wilderness-protected area on long-term vegetation change in the mountains of the Cape Floristic Region, a global biodiversity hotspot

4.1 Introduction

This section of the thesis uses before-after comparisons and counterfactual thinking to investigate the impact of a wilderness-protected area (the treatment), the Groot Winterhoek Wilderness Area (see 2.1.3), on vegetation cover and composition (the outcome) over the last ~40 years in a Cape Floristic Region mountain catchment important for regional water supply. Change in vegetation cover is determined using 73 photographs and 72 vegetation surveys repeated ~40 years after protected area establishment. Causal links between the presence of the wilderness-protected area and protected area outcome are determined by interrogating moderators and mechanisms that may have shaped these links.

Specifically, the influence of climate as a potential moderator of protected area outcome, and interactions with mechanisms such as land use and fire, are discussed drawing on social accounts of land use/cover change from Section 3 and an analysis of land use/cover change from orthorectified aerial images in Section 5. Protected area impact is described in terms of the difference observed between the conditions in the protected area and estimates of the conditions of the same area were protection not present since the 1970s (the counterfactual conditions). The study focuses on the following four questions.

1) How have the conditions in the protected area changed over the last ~40 years in terms of vegetation cover and composition (protected area outcome)?

2) Can changes be related to the establishment of the wilderness-protected area when considering moderating variables not influenced by protected area establishment (e.g. climate) and interactions with mechanisms such as land use and fire?

3) What is the protected area impact when the results are compared with potential counterfactual conditions in the absence of protected area establishment?

4.2 Methods

4.2.1 Study region and area

This study focuses on the Groot Winterhoek Wilderness Area (see 2.1.3 and Figure 4.1).

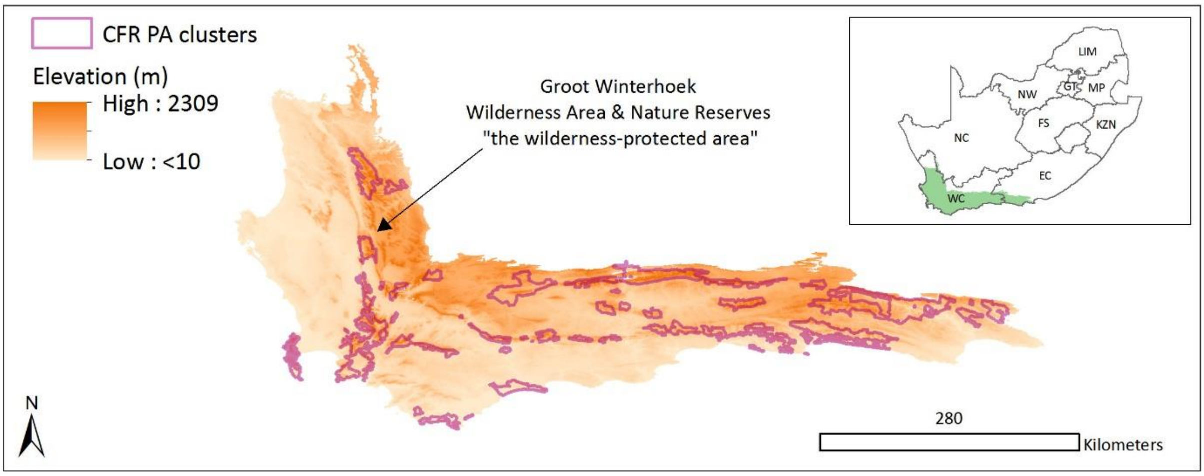


Figure 4.1 Protected area clusters (UNESCO 2015; DEA 2016) and associated elevations (m) (NGI-DEM 2016) in the Cape Floristic Region. The inset map shows South Africa with its provinces (MDB 2013) and the boundary of the Cape Floristic Region in green (SANBI 2012).

4.2.2 Field methods

All historical sites were specifically selected to reflect locations representative of the area.

4.2.2.1 Vegetation surveys

Vegetation surveys, which were originally conducted between 1971 and 1978 were repeated in 2014 and 2015 to assess change in canopy and basal vegetation cover and growth form composition. Vegetation cover intercepts and strikes were measured respectively using line intercept and descending point methods. Sampling procedures used were dictated by descriptions of sampling methods conducted in the 1970s (Table 4.1). The location of sampling sites in the wilderness-protected area is shown in Figure 4.2.

Table 4.1 Sampling procedures based on descriptions of sampling methods conducted in the 1970s.

Year	Sampling method and variables measured	Growth forms measured	Source
Initial survey: **1978** Repeat survey: **2015**	Sixty-one 25 m transects sampled within seven sites using the descending point method. This included recording canopy and basal strikes at 1 m intervals. All transects were in fynbos vegetation. Site location was consistent between years but transects within sites were randomly selected.	Canopy cover was divided into four growth forms: herbs (all non-woody, low growing shrubs especially geophytes and annuals); grasses (Poaceae species); restioids-sedges (Restionaceae and Cyperaceae species); and shrubs (all woody, low growing and tall shrubs). Growth forms were not differentiated for basal intercepts.	(Bands 1985) (D Bands, personal communication, March 2015)
Initial survey: **1971-1973** Repeat survey: **2014**	Eleven permanent 10 m transects sampled using the line intercept method. All basal vegetation intercepts (mm) were recorded. Six transects were in fynbos vegetation and five transects were in outcrop thicket-forest vegetation. Canopy cover was not measured consistently in fynbos sites in the 1970s and was therefore not included when the sites were resampled in 2015.	In certain cases, species were recorded but, in most cases, basal cover was determined for genera or families. This information was used to divide the results into the four growth forms used above: herbs, grasses, restioids-sedges, and shrubs. An extra growth form (trees) was included for certain sites in which typical thicket-forest outcrop tree species were recorded e.g. *Brachylaena* sp..	Jonkershoek Forestry Research Centre (CSIR 1970)

4.2.2.2 Repeat land based oblique photographs

Seventy-three land based, oblique photographs originally taken between 1970 and 1978 (CSIR 1970; Bands 1985) were repeated between 2013 and 2016. Repeat photograph locations were identified and photographs were retaken following procedures summarised by rePhotoSA (2017). Eleven photographs were linked to six of the fynbos line intercept transects and 13 were linked to five of the outcrop thicket-forest line intercept transects. The rest of the photographs were randomly taken throughout the protected area between 1970 and 1978 by different photographers (Figure 4.2).

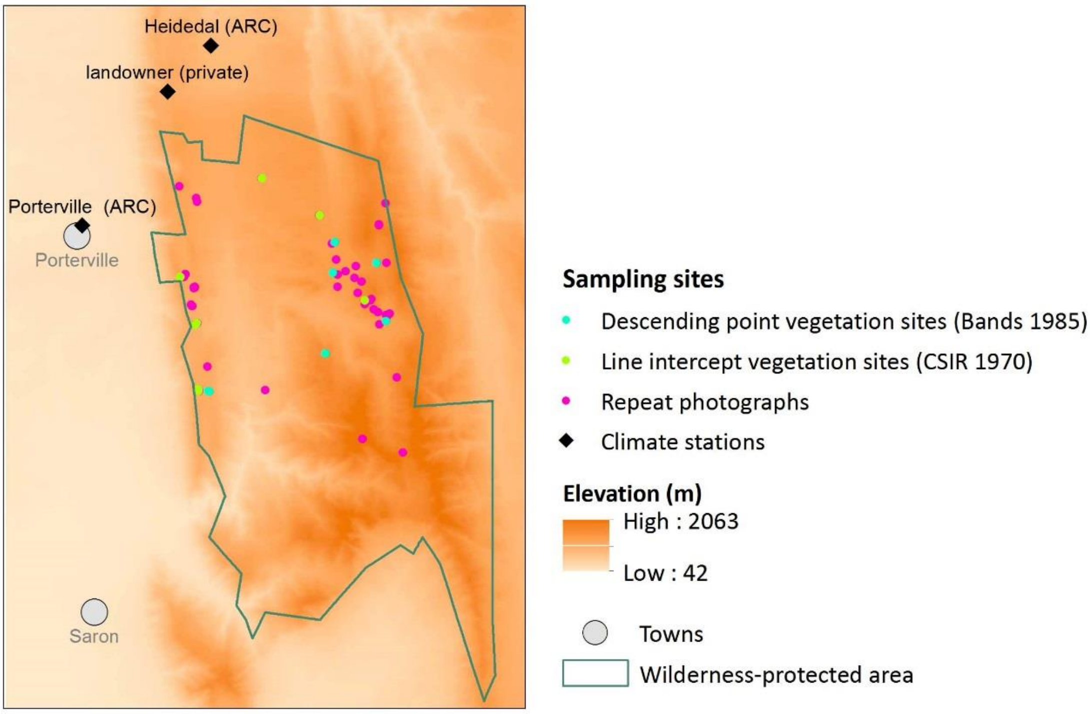


Figure 4.2 Vegetation sampling and repeat photograph sites in the wilderness-protected area (Groot Winterhoek Wilderness Area and Nature Reserves) (DEA 2016) and climate stations nearby used for climate and rainfall analyses. Associated elevations are also shown (NGI-DEM 2016). Descending point sites were originally surveyed by Bands (1985) in 1978 and resurveyed in 2015. Line intercept sites were originally surveyed by researchers from the Jonkershoek Forestry Research Centre (CSIR 1970) between 1971 and 1973 and resurveyed in 2014 (Table 4.1). Repeat photographs were taken between 1970 and 1978 and repeated during 2013 – 2016. Please refer to Figure 4.1 for the wider geographical area within which the Cape Floristic Region is apart.

4.2.3 Data and statistical analyses

All statistical analyses were performed in R (R Core Team 2016).

4.2.3.1 Descending point vegetation data analysis

A generalised linear mixed model with binomial distribution and Laplace approximation was performed to determine change in the proportion of canopy and basal vegetation cover between the two sampling dates[10]. Year of sampling was modelled as the fixed effect. Transects were nested within sites and modelled as random effects (Table 4.2). Model fit and

[10] R package: lme4; function: glmer (Bates 2010; Bates et al. 2015a, 2015b)

over-dispersion was assessed by comparing the sum of the squared Pearson residuals to the residual degrees of freedom followed by residual analysis[11] (Bolker et al. 2009; Zuur et al. 2009).

Table 4.2 Generalised linear mixed model variables and descriptions, used for modelling the change in the proportion of canopy and basal vegetation cover over the period 1978-2015.

Variable	Type	Description
Year	Fixed	1) **1978:** 7 years prior to the formal proclamation of the wilderness-protected area and in which the last sections of land were expropriated from landowners. 2) **2015:** 37 years after expropriation of the last private properties in the area and 30 years after the formal proclamation date of the wilderness-protected area.
Vegetation cover	Response	Binomial counts: Number of canopy and basal intercepts per transect out of a total of 25 potential intercepts situated at 1 m intervals along each transect.
Sites	Random	Seven fixed sites sampled in 1978 and resampled in 2015. Sites were equivalent in terms of their post fire successional maturity (i.e. sampled at 4-6 years post fire) (Bands 1985) (D Bands, personal communication, June 2016). The post fire age prior to the most recent burn in 2009 was 14 years (CapeNature 2016b).
Transects	Random	61 random transects sampled in 1978 across the seven fixed sites and resampled in 2015. Sample size per site was fixed and ranged from 4 to 20 transects depending on the site (see Results Figure 4.4).

Pearson's chi square and pairwise post hoc chi square tests were used to assess associations between growth form composition and sampling year as well as to determine associations between canopy and basal layer compositional proportions for 2015 sampling results. Fisher's exact test and pairwise comparisons were used when expected frequencies were small. Hybrid approximation for Fisher's exact test was used for larger than 2 x 2 tables. Bonferroni corrections were applied for all multiple comparisons[12] (Clarkson et al. 1993; MacDonald & Gardner 2000). A principle component analysis (PCA) was used to visualise vegetation compositional shifts for all transects sampled in 1978 and resampled in 2016 transects[13].

4.2.3.2 Line intercept vegetation data analysis

The Wilcoxon paired signed rank test was used to compare changes in basal vegetation cover, and changes in the cover of shrubs and restioids-sedges, measured using permanent line intercept transects. The smaller sum of the positive ranks and sum of the negative ranks is presented as the test statistic[14]. Because of numerous zeros recorded for grasses, herbs and trees, the results were grouped into categories and the Fisher's exact test was used to compare the changes in the proportion of zeros and abundance categories.

4.2.3.3 (Dis) similarities between vegetation sampling and repeat photograph sites

Agglomerative hierarchical clustering was performed using Gower's distance (Pavoine et al. 2009) and Ward linkages to determine (dis) similarities between vegetation sampling sites, as well as (dis) similarities between repeat photograph locations, in terms of biophysical

[11] R package: stats, blmeco; functions: dispersion_glmer (Korner-Nievergelt et al. 2015); overdisp_fun (Bolker 2009)
[12] R package: stats, function: chiq.test, fisher.test
[13] R package: FactoMineR; functions: PCA (Lê et al. 2008a, 2008b)
[14] R package: stats; function: wilcox.test

characteristics. The number of groups were determined by evaluating the largest difference of heights between nodes in combination with using a variable cut height approach for interpreting finer scale clusters[15]. Biophysical characteristics were extracted for each vegetation-sampling site and repeat photograph location using ArcGIS and a range of data sources (ESRI 2015) (Table 4.3). Where appropriate, site information was extracted per photograph at the point at which the photograph was captured. Where photographs represented areas further afield from the point of capture an additional point was inserted at the correct location and this was used for extracting information. Two photographs were excluded from the clustering procedure as they were at a landscape level and the appropriate level of detail required for an analysis of vegetation change could not be determined.

Table 4.3 Biophysical site characteristics and sources used for agglomerative hierarchical clustering of vegetation sampling and photograph sites.

Site variable	Description	Source
Aspect	i) east; ii) northeast; iii) southeast; iv) southwest; or v) west	Photogrammetrically corrected digital elevation model[16] (NGI-DEM 2016)
Slope	0-90 degrees	
Altitude	elevation above sea level in meters	Hand-held GPS
Soil	i) sandstone-quartzitic derived; or ii) shale-derived (this includes partly shale-derived soils near the shale-band).	Georeferenced local soil map (Bands 1985)
Geology	i) sandstone minor grit, conglomerate and shale; ii) quartzitic sandstone with thin shale and conglomerate lenses; iii) thinly bedded sandstone, siltstone and mudstone; or iv) shale and arenaceous shale.	Georeferenced local geology map (Bands 1985)
Vegetation	i) Winterhoek Sandstone fynbos (FFs 5); ii) Northern Inland Shale-Band fynbos (FFb 1); or iii) Western Altimontane Sandstone Fynbos (FFs 30).	(Mucina & Rutherford 2011; SANBI 2012) and current 2014 orthoimages.
Local vegetation	i) fynbos; or ii) thicket-forest (i.e. referred to as fynbos thicket in Mucina & Rutherford (2011) but not specifically mapped).	Historical and repeat ground photos (CSIR 1970; Bands 1985)
Unique local characteristics	i) rocky outcrop; ii) rocky peak; iii) rocky or foot slope; iv) hydromorphic; v) shale-band; or vi) valley bottomland	Field descriptions from 1970-1978 and 2013-2016; georeferenced local sensitive sites mapping (CSIR 1970; Bands 1985) and current 2014 orthoimages.

4.2.3.4 Repeat photograph vegetation change analysis

Matched historical and repeat photographs were georeferenced to overlay directly over historical images. A fishnet grid of 750 cells was used to sample the matched images[17]. The following classes were applied: vegetation, bare ground, road, and rock. A point marker was placed in the centre of each cell and the specific class was recorded on both images. Only foreground areas that were representative of homogenous direct biophysical characteristics used for agglomerative clustering were classified using the above procedure. Two images

[15] R packages: cluster, dynamicTreeCut; functions: daisy, hclust, cuttreeDynamicTree (Langfelder et al. 2008, 2016; Maechler et al. 2016)

[16] ArcGIS spatial analyst tools: surface: aspect and slope (ESRI 2015)

[17] ArcGIS georeferencing toolbar, fit to display; ArcGIS data management tool: create fishnet (ESRI 2015)

were not sufficiently matched for quantitative comparison and therefore were removed from the analysis. A further four images were removed due to poor historical photograph quality (i.e. total n = 67). Please refer to Boyer et al. (2010) for more information on the constraints and challenges associated with comparing historical and repeat photography. The Wilcoxon paired signed rank tests with a continuity correction in the normal approximation for the p value was used to compare overall change in cover classes as well as change relevant to specific coarse-scale groupings of photographs based on agglomerative clustering results (see 4.2.3.3 above). Bonferroni corrections were used to adjust p values for post hoc comparisons[18].

To provide context to the quantitative analysis, and to capture further details on change in growth forms and species, a qualitative approach was used. This was supported by NVivo qualitative data analysis software (NVivo 2012), and ArcGIS (ESRI 2015). The approach included visually assessing high-resolution photograph sets in ArcGIS and determining species or growth forms which could be adequately discerned between matched historical and repeat photographs. Qualitative descriptions of change for specific growth forms and cover types were captured per photograph using NVivo. Photograph descriptions were then coded into two main themes of change: i) an increase; or ii) a decrease, which further consisted of sub-themes related to cover types, namely: i) grasses; ii) reseeding proteoids; iii) resprouting proteoids; iv) restioids; v) non-proteoid shrubs; vi) *Stoebe* spp.; and vii) thicket-forest vegetation. Themes were determined using an inductive approach and based on descriptions generated for each photograph set. The resulting coded themes were classified against coarse-scale groupings identified using agglomerative clustering (in Section 4.2.3.3) i.e. outcrop thicket-forest, shale fynbos and sandstone-quartzitic fynbos. Classified frequencies of the total number of photographs coded per theme and sub-themes were exported and analysed, using Pearson chi square and Fischer exact tests, to determine whether the distribution of photographs coded for specific cover types differed within and between coarse-scale groupings[19].

4.2.3.5 Spatial autocorrelation

The Mantel's test was performed to assess the relationship between the similarity between sites in terms of biophysical variables used in agglomerative clustering and the geographic distance between repeat photograph and vegetation survey sites. It was also used to assess whether any spatial autocorrelation existed between changes in vegetation, bare ground, rock or road cover observed at repeat photograph sites and basal and canopy cover observed at vegetation survey sites[20] (Legendre & Fortin 2010). There was no spatial autocorrelation between change in cover types and the geographic distance between repeat photograph and vegetation survey sites with the Mantel statistic r for all cover types ranging between -0.14 and 0.16. Mantel statistic r and p-values for repeat photographs were: i) bare: r = -0.06, p =

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[18] R package: stats; function: wilcox.test
[19] R package: stats; function: chiq.test, fisher.test
[20] R package: vegan, cluster; function: mantel, daisy (Maechler et al. 2016; Oksanen et al. 2017)

0.96; ii) rock: r = 0.029, p = 0.2, iii) road: r = -0.001, p = 0.49; and iv) veg: r = -0.03, p = 0.84. For the vegetation survey sites, these were: i) fynbos basal: r = 0.14, p = 0.11; ii) fynbos canopy: r = -0.14, p = 0.65; iii) forest basal: r = -0.09, p = 0.63; and iv) forest canopy: r = -0.16, p = 0.6.

Lumped biophysical site characteristics were moderately correlated with the distance between individual photographs and vegetation survey sites. For example, photographs and vegetation survey site pairs further away from each other were more dissimilar in terms of biophysical characteristics than photographs and survey sites closer to each other. However, when considering biophysical characteristics individually, there was an overall very weak to weak effect of the distances between paired sites on the (dis) similarity between sites (Mantel statistic r being between -0.09 and 0.31). One exception was altitude which showed moderate correlation coefficients with the distance between paired sites in all cases (Mantel statistic r = 0.51 – 0.51) (Table 4.4). To provide an overview of the association between agglomerative clustering results and the spatial distance between repeat photographs and vegetation survey sites, maps are included in 4.3.3 and 4.3.4.1.

Table 4.4 Mantel's test correlation coefficients (r) and p values shown in brackets for associations between (dis) similarity in biophysical variables used in agglomerative clustering and the geographic distance between repeat photograph and vegetation survey sites.

Biophysical characteristics	Repeat photographs (n=71)	Repeat photographs (n=67)	Repeat vegetation sites (n=18)
Lumped characteristics	r = 0.45 (0.0001)	r = 0.43 (0.0001)	r = 0.42 (0.02)
Aspect	r = 0.09 (0.001)	r = 0.09 (0.001)	r = 0.23 (0.008)
Slope	r = 0.11 (0.001	r = 0.11 (0.001)	r = 0.18 (0.03)
Altitude	r = 0.59 (0.0001)	r = 0.58 (0.0001)	r = 0.51 (0.02)
Soil	r = 0.19 (0.001)	r = 0.17 (0.0001)	r = 0.27 (0.12)
Geology	r = 0.17 (0.001)	r = 0.19 (0.0001)	r = 0.31 (0.004)
Vegetation	r = 0.16 (0.0003)	r = 0.11 (0.005)	r = -0.03 (0.67)
Local vegetation	r = 0.27 (0.001)	r = 0.29 (0.0001)	r = 0.17 (0.04)
Unique characteristics	r = 0.22 (0.001)	r = 0.21 (0.0001)	r = 0.29 (0.003)

4.2.3.6 Climate history

Total monthly rainfall (mm), mean monthly temperature (°C), total monthly A-pan evaporation (mm) and average daily wind run (km.day) observations from 1974 to 2016 were available for a lowland weather station (Porterville: 33.01247 S and 18.99947 E, altitude 145 m), ~13 km from the centre of the wilderness-protected area (Figure 4.2). Monthly rainfall data from 1977 to 1984 were also available for a mountain station (Heidedal: -32.93333435 S and 19.06666756 E, altitude 745 m), ~17 km from the centre of the wilderness-protected area. These data were obtained from the Agricultural Research Council. In addition to these records, total monthly rainfall data for the years 1991 to 2016 were obtained from a landowner's rainfall gauging station situated on the mountain and at an altitude 843, ~2 km from the Heidedal station and ~15 km from the centre of the wilderness-protected area. To develop a rainfall record for the "mountains" the mean monthly difference between the mountain station data and the lowland Porterville station were calculated. The missing years of monthly rainfall for the mountains were estimated by adding the mean monthly difference to the Porterville station data for the years 1974-1976 and 1985-1991. The Porterville station was ~8 and ~11 km from the Heidedal and landowner station respectively.

Observed trends in annual and seasonal total rainfall (mm), and mean maximum and minimum temperature (°C), A-pan evaporation (mm), and wind run (km/day) were evaluated using the non-parametric Mann-Kendall trend test. The correlation coefficient tau produced by the test provides an indication of the relative strength and direction of the trend. The slope of the trend was calculated using the Theil–Sen estimator, which is the median of the slopes calculated between all pairs of data points in the series[21]. Trend estimates were calculated for seasonal and annual rainfall totals and climate means considering a winter rainfall hydrological year, April to March, and the following seasons: summer (December, January, February), autumn (March, April, May), winter (June, July, August), and spring (September, October, November). A smoothed curve was added to figures of annual totals and means using a Loess filter[22].

4.2.3.7 Fire history

Post fire successional maturity equivalence between historical and repeat vegetation survey sites and photographs was assessed using several approaches, including:

i) a spatial fire dataset (CapeNature 2016b)[23];
ii) personal communication (D Bands, pers.comm., 20 June 2016);
iii) visual assessment of historical and repeat ground photographs (CSIR 1970); and
iv) information on expropriation dates from the records of the Deeds office in Cape Town combined with documentary evidence where available on burning occurrence prior to expropriation dates (Bands 1985).

The reason for the use of more than one approach was that there were no fire records in the spatial fire dataset prior to 1974 for the wilderness-protected area and there were only two fire records prior to 1978. These fires occurred in 1975 and 1976 but only burnt very small portions of the wilderness-protected area. Furthermore, fire records in the dataset were inherently unreliable and patchy for 1974-2000. For example, many records in the dataset had an indication that the date was not certain i.e. “START DATE FAKE” (T Forsyth, personal communication, 15 February 2017). This was supported by a recent regional fire analysis, which used the spatial fire dataset for nearby protected areas in the mountains of the Cape Floristic Region but which excluded the wilderness-protected area due to unreliable data (van Wilgen et al. 2010). Despite fire records being unreliable for 1974 – 2000 in the spatial fire dataset (CapeNature 2016b), the number of fires occurring since 1974 for the wilderness-protected area is shown in Figure 4.3. Vegetation age considerations for historical and repeat vegetation survey and photograph analyses are discussed separately below.

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[21] R package: trend functions: mk.test, sens.slope
[22] R package: ggplot2, functions: geom_smooth (method = loess)
[23] ArcGIS spatial analyst tools: extract multi values to points

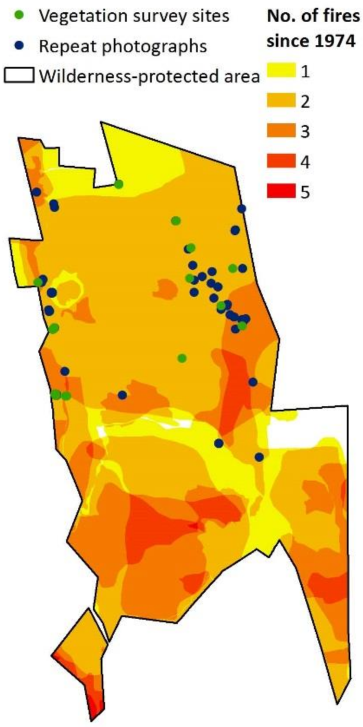

Figure 4.3 Number of fires occurring at repeat vegetation sampling and photograph sites in the wilderness-protected area since 1974 (CapeNature 2016b).

4.2.3.6.1 Repeat descending point and line intercept vegetation surveys

It was possible to use the spatial fire dataset to determine the post fire age for all repeat surveys conducted. This is because a fire occurred in 2009 which burnt all transects. Thus, vegetation age for all repeat descending point and line intercept vegetation surveys was 6 and 5 years respectively.

4.2.3.6.2 Historical descending point vegetation surveys

For historical descending point vegetation surveys, undertaken in 1978, an estimate of four to six years was provided for the vegetation age by the original surveyor (D Bands, personal communication, June 2016). It was indicated that every attempt was made to work in vegetation greater than 4 years for all sites and that vegetation of greater than 6 years of age was rare for the sites surveyed especially for the vegetation associated with the shale-band and partly shale-derived soils. This was due to frequent burning being applied to the land as a management tool in the past prior to expropriation (D Bands, personal communication, June 2016). Estimates of 4-6 years were largely comparable to the 6 years post fire vegetation age estimated for sites that were repeated using descending point surveys in 2015. Therefore, the historical and repeat descending point sites were assumed to be relatively equivalent in terms of post fire successional maturity.

4.2.3.6.3 Historical line intercept vegetation surveys

For the line intercept surveys, vegetation ages were not available as fire dates were not recorded for the original photographs (CSIR 1970) and interviews were not possible with the original surveyors. Given the issues associated with determining fire ages from the spatial fire data set described above, visual comparisons between historical and repeat photographs, which existed for each of these sites, was used to evaluate post fire successional equivalency. In addition, the year of expropriation of specific farm portions was used as an indication of the last potential year that the area could have been burnt deliberately. Based on a visual comparison of historical and repeat photographs, it was likely that the vegetation was of similar age between historical and repeat line intercept sites. Furthermore, there was only a moderate relationship between the difference in vegetation cover and the difference in vegetation age between historical and repeat line intercept surveys when age was determined using expropriation dates and the spatial fire data set for historical and repeat photographs respectively (Kendall's $\tau = 0.49$; $p = 0.055$)[24]. As a result, line intercept sites were assumed relatively equivalent in terms of post fire successional maturity.

4.2.3.6.4 Historical and repeat photographs

The post fire age of the vegetation in 64 of the repeat photographs ranged from four to six years as determined from the fire spatial dataset. Nine photographs had not burnt post 2000 and based on records in the fire spatial dataset prior to 2000 the post fire age of four and five of these photographs respectively was 18 and 15 years. No fire data existed prior to 1978 for 63 photograph locations. Given the lack of a reliable fire history for the study area a visual assessment was used to assess whether photographs were adequate in terms of comparisons given the potential post fire maturity conditions. Based on a visual comparison of historical and repeat photographs, it is likely that the vegetation was of similar age between historical and repeat photographs. There was also no relationship between the difference in vegetation cover and the difference in vegetation age between historical and repeat photographs when age was determined using expropriation dates and the spatial fire data set for historical and repeat photographs respectively (Kendall's $\tau = -0.14$; $p = 0.1$)[25]. This further supported the visual assessment that photographs were relatively equivalent in terms of post fire successional maturity.

4.3 Results

4.3.1 Descending point sites 1978 - 2015

4.3.1.1 Canopy and basal fynbos vegetation cover in 1978 and 2015

Canopy and basal vegetation cover increased in the fynbos sites resampled using the descending point method (Figure 4.4). Increases in canopy cover ranged from 8% to 38% with a mean increase of 24% and from 19% to 47% for basal cover with a mean increase of 30%. The odds of striking vegetation in 2015 in comparison to 1978 were 4.5 and 3.6 times higher respectively in the canopy ($\beta = 1.5 \pm 0.1$, $z = 14.99$, $p<0.0001$) and basal layer ($\beta = 1.29 \pm 0.07$, $z = 16.46$, $p<0.0001$).

[24] R package: stats; function: cor.test
[25] R package: stats; function: cor.test

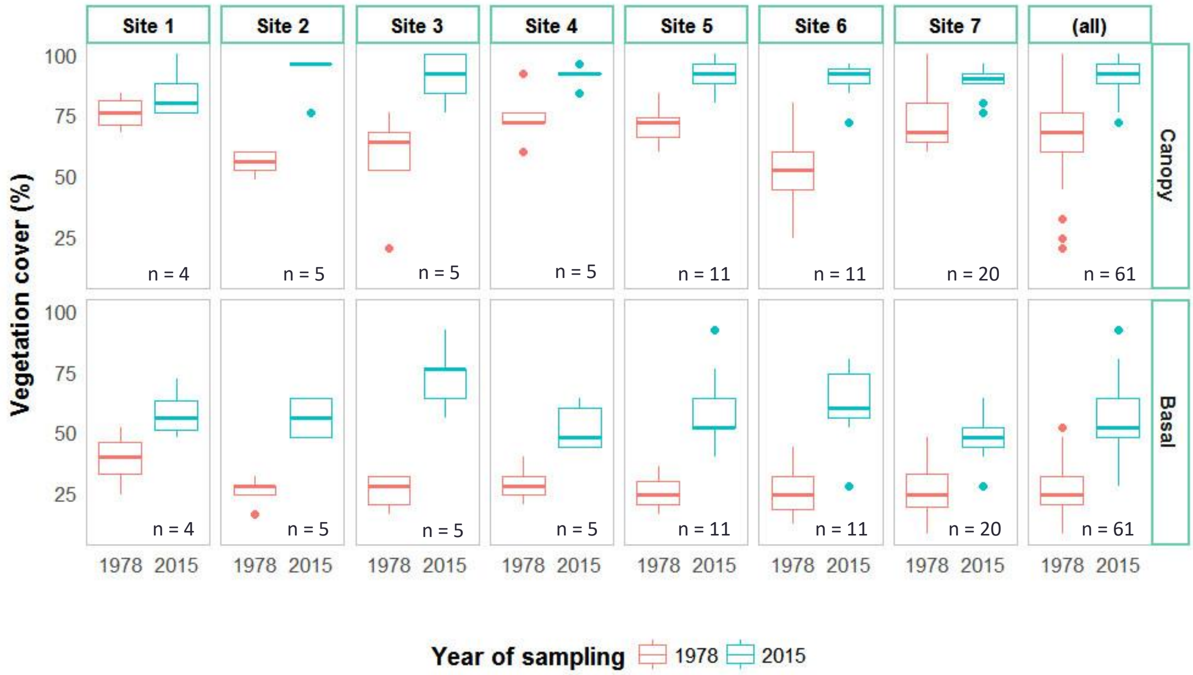

Figure 4.4 Change in canopy and basal vegetation cover sampled in 1978 and then resampled in 2015. Sites surveyed were relatively equivalent in terms of their post fire successional maturity (see Section 4.2.3.7).

4.3.1.2 Vegetation compositional changes

There was an association between the year of sampling and the pooled frequency of observed growth forms and bare ground in the canopy vegetation ($\chi 2$ = 567.15, df = 4, p-value<0.0001) (Figure 4.5). The most important contributing factors to the definition of the $\chi 2$ score were shrubs (contribution: 48%), bare ground (contribution: 33%) and restioids-sedges (contribution: 11.4%). There was an inverse relationship between the Pearson residuals for growth form types and bare ground between 1978 and 2015.

For example, 1978 was positively associated with bare ground, restioids-sedges and herbs and 2015 was positively associated with grasses and shrubs and negatively associated with bare ground, restioids-sedges and herbs (Figure 4.5). Multiple comparisons for growth forms showed associations between cover and sampling date for grasses ($\chi 2$ = 37.11, df = 1, p<0.001), restioids-sedges ($\chi 2$ = 96.032, df = 1, p<0.001), shrubs ($\chi 2$ = 396.61, df = 1, p<0.001) and bare ground ($\chi 2$ = 241.2, df = 1, p<0.001). There were no associations between year of sampling and herb cover ($\chi 2$ = 4.41, df = 1, p = 0.15).

Principle component analysis confirmed the results above and separated the 1978 sites from the 2015 sites along the first principle component, which was positively correlated with shrubs and negatively correlated with restioids-sedges. The 2015 sites were positively correlated with shrubs and grasses while the 1978 sites were positively correlated with restioids-sedges with some overlap for specific transects (Figure 4.6 and Table 4.5)

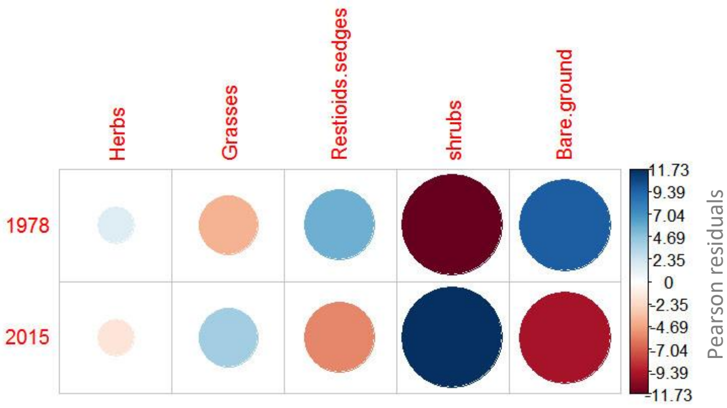

Figure 4.5 Pearson residuals for each cell. Positive (negative) values are in blue (red) and specify a positive (negative) association between corresponding year (row) and growth form or bare ground (column). The size of the circle reflects the residuals$^2/\chi2$, which is an indication of the contribution of each growth form type as well as for bare ground to the overall $\chi2$ score.

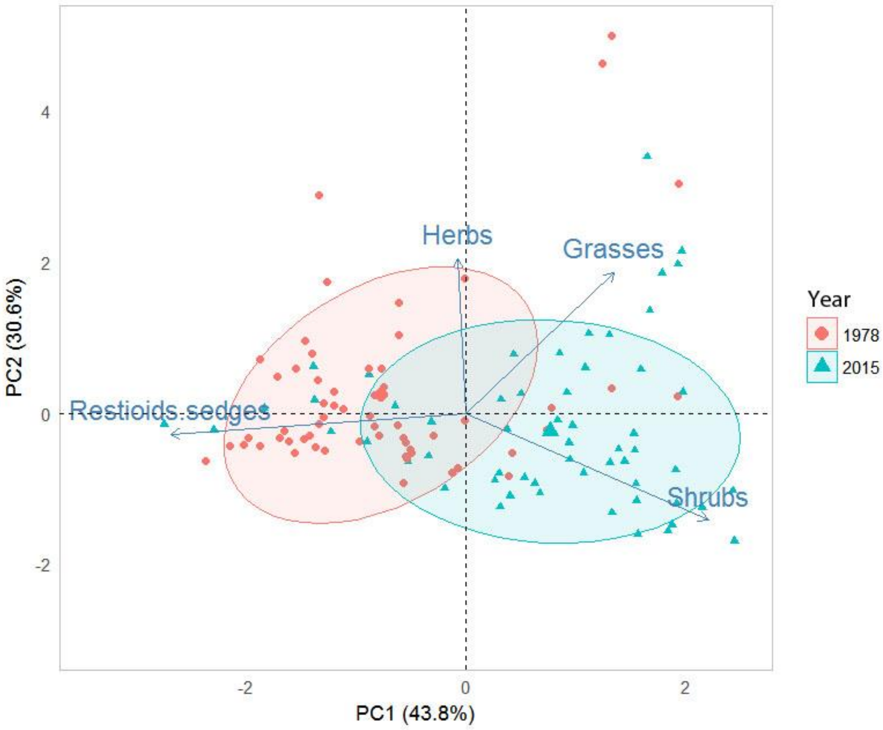

Figure 4.6 Principle component analysis results indicating deviations between transects surveyed in 1978 and resurveyed in 2015 based on growth form composition.

Table 4.5 Contributions (%) of individual growth forms to the dimensions of the principle component analysis, PC1 and PC2, shown in Figure 4.6.

Growth form contributions (%)	PC1	PC2
Herbs	0.03	42.9
Grasses	13.2	35.6
Restioids-sedges	51.8	0.7
Shrubs	35.0	20.6

For single site comparisons, all sites except one showed deviances from the 1978 growth form distributions (*Fisher's exact p-value*: site 1 = 0.06; site 2, 3, 4, 5, and 6 < 0.001; site 7: $\chi 2$ = 353.46, df = 1, p = 0.001) (Figure 4.7). Five individual sites contributed to the overall trend of an increase in shrubs since 1978 ($\chi 2$ = 140.41 [site 3]; $\chi 2$ = 76.34 [site 4]; $\chi 2$ = 13.09 [site 5]; $\chi 2$ = 36.734 [site 6]; $\chi 2$ = 254.72 [site 7]; df = 1, p<0.001). However, only three of these sites contributed to the reduction in restioids-sedges ($\chi 2$ = 55.46 [site 3]; $\chi 2$ = 37.66 [site4]; $\chi 2$ = 191.06 [site 7], df = 1, p<0.0001). Two sites showed an opposite trend including an increase in restioids-sedges (site 2: $\chi 2$ = 12.235, df = 1, p<0.001; site 6: $\chi 2$ = 8.07, df = 1, p = 0.004). Three sites contributed to the increase in grasses observed ($\chi 2$ = 13.3 [site 2]; $\chi 2$ = 14.48 [site 5]; $\chi 2$ = 16.955 [site 7], df = 1, p<0.001).

There was no change in the distribution of herbs between sampling dates for all sites (Fishers exact p > 0.05). Growth form composition was dependent on the layer in which it was sampled for the sites sampled in 2015 (Fisher's exact p value: site 1-6 p <0.0001; site 7: $\chi 2$ = 210.91, df = 4, p<0.0001). Most sites had greater shrubs in the canopy layer in comparison to the basal layer ($\chi 2$ = 7.66, df = 1, p =0.0056 [site 1]; $\chi 2$ = 22.231 [site 3]; $\chi 2$ = 32.4 [site 4]; $\chi 2$ = 18.22 [site 5]; $\chi 2$ = 43.20 [site 6]; $\chi 2$ =131.98 [site 7], df =1, p<0.0001). However, one site had greater restioids-sedges in the canopy layer in comparison to the basal layer (site 2: $\chi 2$ = 22.317, df = 1, p<0.0001).

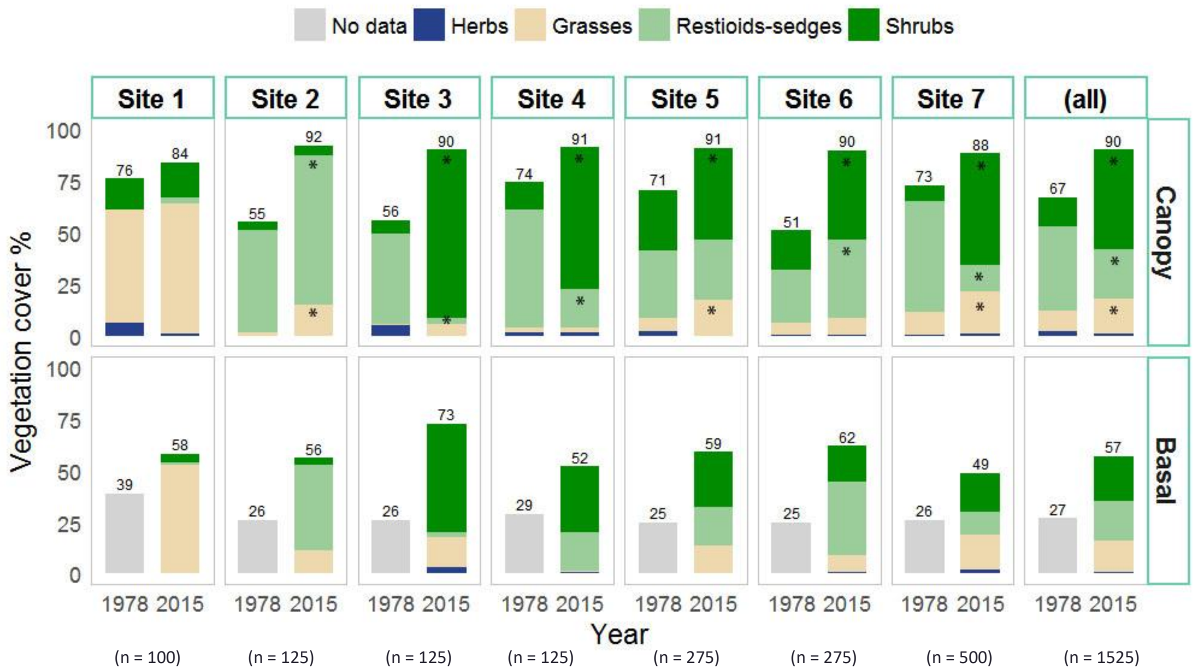

Figure 4.7 Observed frequencies of growth forms sampled at points (n) along transects in 1978 and again in 2015 at seven sites (1-7) and aggregated for all points sampled (all). In 1978, only canopy cover (and not basal cover) was recorded for individual growth forms. Asterisks are used to denote p values < 0.05 for multiple post hoc comparisons for each growth form type between sampling dates in the respective growth form in 2015.

4.3.1.3 Species level influence

Species contributing most to the overall canopy cover at the sites included *Aspalathus* sp. (13.8%), *Protea repens* (11%), *Elytropappus glandulosus* (10%), *Stoebe plumosa* (6.6%), *Geochloa rufa* (5.9%) and *Aristida* spp. (4.6%). Species contributing most to the overall basal cover included *Geochloa rufa* (8.3%), *Restio* spp. (7.3%), *Stoebe plumosa* (7.2%), *Aristida* spp. (6.8%), *Elytropappus glandulosus* (6%), and *Protea repens* (5%).

4.3.2 Paired permanent transects 1970 - 2014

4.3.2.1 Fynbos basal vegetation cover and composition

Basal cover increased in fynbos sites resampled using the line intercept method (V = 0, p = 0.031). The increase ranged from 10% to 33% with a mean increase of 16%. There was a difference in restioids-sedges in the basal vegetation cover (V = 0, p = 0.031). However, there were no clear differences between the cover of herbs, grasses, shrubs and trees in the basal layer (herbs: *Fischer's exact p* = 0.54; grasses: *Fischer's exact p* = 0.24; shrubs: V = 2, p = 0.093; trees: *Fischer's exact p* = 1). This is an indication that the increase in restioids-sedges was responsible for the overall increase in basal vegetation cover (Figure 4.8 and see Table 4.6, Figure 4.9 to Figure 4.13, for the historical and repeat photograph sets for each of the sites sampled).

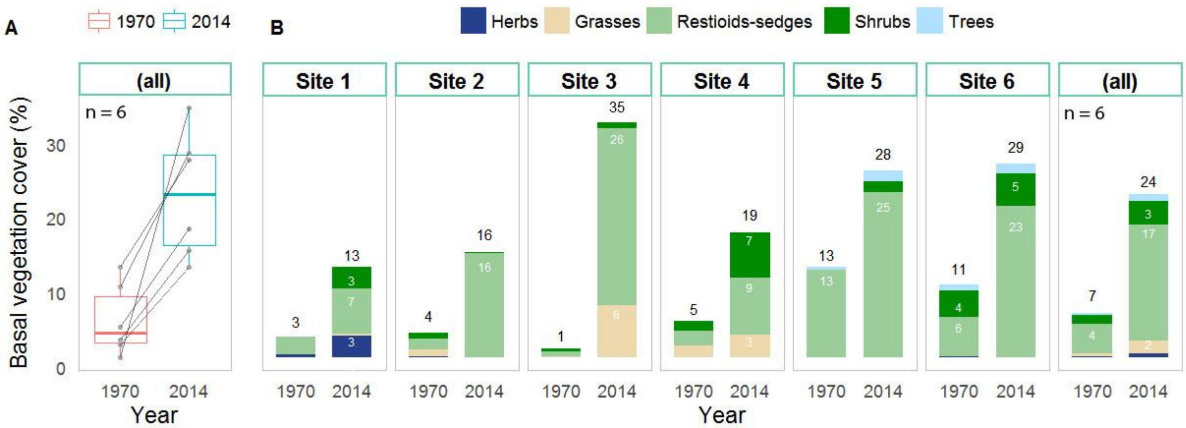

Figure 4.8 A) Changes in basal vegetation for paired transects, B) Changes in basal vegetation cover of different growth forms for paired transects. Historical and repeat photograph sets for each site sampled are shown in Table 4.6, Figure 4.9 to Figure 4.13.

Table 4.6 Repeat photographs for each sampling transect (CSIR site 1 – 6). The increase in vegetation cover of restioids-sedges is evident in all photograph sets. Other notable changes include decreased bare areas in sites 1-2, decreased exposure of rock in sites 4-6, overall increase in height of vegetation in all photographs, loss of taller reseeding *Protea laurifolia* and resprouting *Protea nitida* individuals in sites 1-2 but notable increases in juvenile *Protea* spp., including *Protea repens*, and a decrease in *Stoebe plumosa* in site 3.

Figure 4.9 **CSIR Site 1:** historical (1970) and repeat (2013) image set. Original photograph: RA Haynes; Repeat: S Jack.

Figure 4.10 **CSIR Site 2:** historical (1970) and repeat (2013) image set. Original photograph: RA Haynes; Repeat: S Jack.

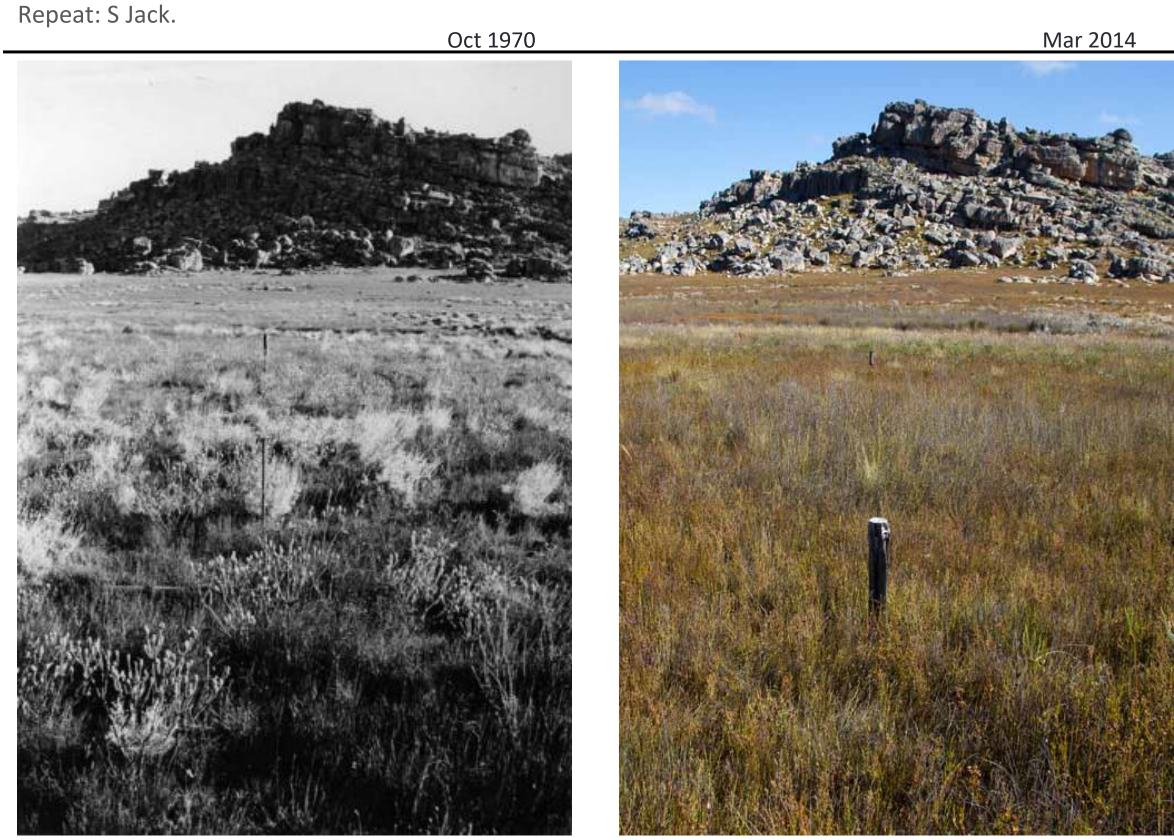

Figure 4.11 **CSIR Site 3:** historical (1970) and repeat (2014) image set. Original photograph: FJ Kruger; Repeat: S Jack.

Figure 4.12 **CSIR Site 4:** historical (1971) and repeat (2014) image set. Original photograph: H Howe; Repeat: S Jack.

Figure 4.13 **CSIR Site 5 and 6:** historical (1971) and repeat (2013) image set. Original photograph: H Howe; Repeat: S Jack.

4.3.2.2 Outcrop canopy and basal vegetation cover

Outcrop thicket-forest sites mostly showed a decline in canopy vegetation cover, but this was not evident at all sites. Changes in canopy cover ranged from a 55% decrease to a 0.5% increase in canopy cover (V = 14, p = 0.13). Basal vegetation cover showed a marginal increase and ranged from 2% to 8% (V = 0, p = 0.063) (Figure 4.14).

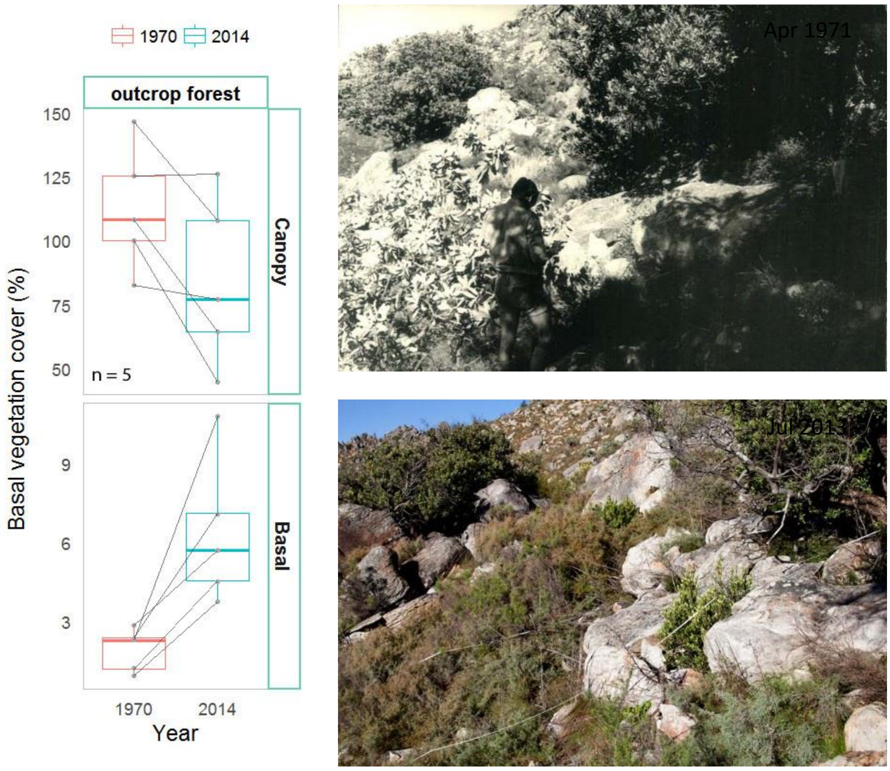

Figure 4.14 Vegetation cover change for the five outcrop forest transects (left). The repeated pair of photographs on the right provide an example of the type of changes which were typically evident at the outcrop forest sites. The photographs show an increase in *Stoebe* sp. basal cover along the thicket-forest margins and a general decline in tree canopy cover. Other notable changes in this repeat include a loss of *Protea nitida* from the forest margin**.** Original photograph: RA Haynes 1971; Repeat: S Jack 2013.

4.3.3 (Dis) similarity in biophysical variables at vegetation sampling sites

The fynbos line intercept and descending point sites clustered into three groups while the thicket-forest outcrop sites clustered together forming a fourth group (Figure 4.15). Each group contained sites with similar biophysical profiles (Table 4.3), except for Group 4, which contained a mixture of sites that were not aligned with the other three groupings. A summary of each group's biophysical profile and results in terms of vegetation change is provided below. The spatial distribution of groups in relation to the distance between individual sites is shown in Figure 4.16.

- **Group 1** (Shale-band) contained four of the descending point sites (Bands site 3, 5, 6, 7). Vegetation cover increases were largely characterised by increases in shrubs. Sites were situated on the shale-band in Northern Inland Shale-Band vegetation and consisted of shale-derived soils with east or southeast facing slopes ranging from 8 to 23 degrees (mean 11 degrees) and a mean altitude of 1122 m (range: 917 – 1298 m).

- **Group 2** (Hydromorphic and one sandstone slope) comprised one descending point site (Bands site 2) and four of the line intercept sites (CSIR sites 3-6). Bands site 2 was the only descending point site that showed an increase in restioids-sedges with shrubs remaining unchanged. All four of the line intercept sites also showed increases in restioids-sedges with no change in the cover of shrubs. Sites were characteristic of hydromorphic areas (except for CSIR 4 which was situated on a rocky slope), with west or southwest aspects, altitudes of >1000 m (range: 1005 – 1262), and sandstone-quartzitic derived soils with a mean slope of 5 degrees (range1.7 – 16.9 degrees).
- **Group 3** (Outcrop thicket-forest) were thicket-forest rocky outcrop sites, which showed no clear directions of change in canopy cover with increases in basal cover from before-after vegetation comparisons.
- **Group 4** (Partly shale slopes and valley bottomland) included the remaining sites that did not necessarily align with the above three groupings. This included two descending point sites: i) Bands site 1, which was largely dominated by grasses in the canopy and basal layer; and ii) Bands site 4, which showed increases in shrubs in the canopy layer, and was characteristic of valley bottomlands and partly shale-derived soils. It also contained two-line intercept sites (CSIR 1 and 2), which showed increases in the cover of predominantly restioids-sedges in the basal layer and were characteristic of rocky slope fynbos vegetation on partly shale-derived soils.

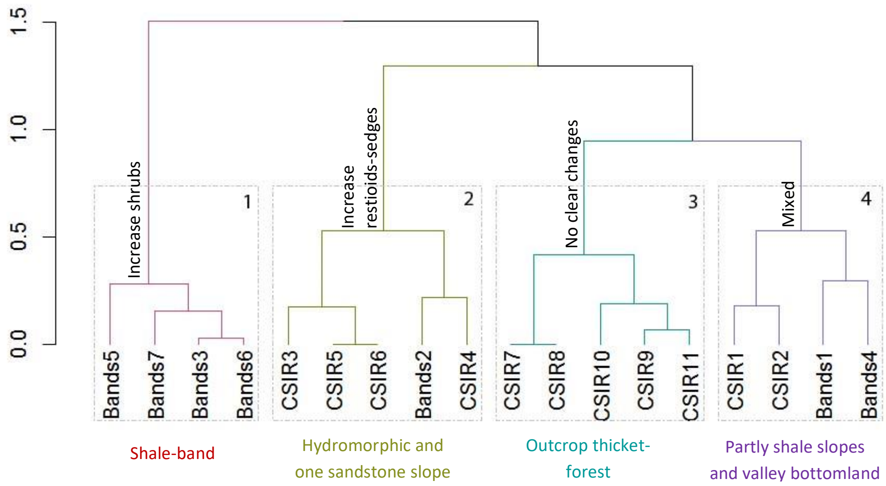

Figure 4.15 Agglomerative hierarchical clustering based on Gower's distance for biophysical site characteristics (aspect, slope, altitude, soils, geology, vegetation and unique site characteristics). Four groupings are evident which correspond broadly to 1) sites with increases in shrubs; 2) sites with increases in restioids-sedges; 3) no change in cover and comprised mostly of thicket-forest outcrop sites; 4) mixture of sites driven by change in shrubs, grasses and restioids-sedges.

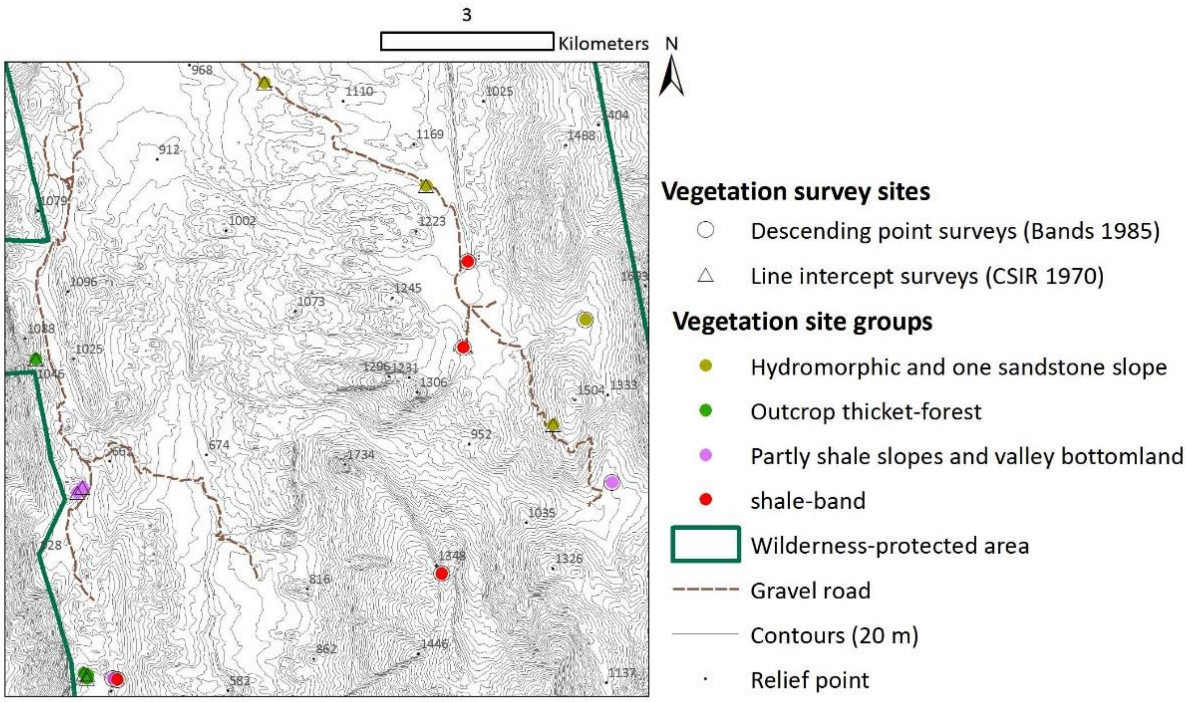

Figure 4.16 Spatial distribution of vegetation survey groups as identified in agglomerative clustering results presented in Figure 4.15 above.

4.3.4 Repeat photos

4.3.4.1 (Dis) similar repeat photographs in terms of biophysical characteristics

Agglomerative clustering resulted in three coarse-scale groups and eight fine-scale groups for repeat photograph sites. At the higher branching level groups were largely driven by a difference between local vegetation and soils and at the lower level by a combination of respective altitudes, slopes, underlying geologies and unique site characteristics. The following includes a description of the groups while Figure 4.17 shows the spatial distribution of coarse- and fine-scale groups in relation to the distance between individual repeat photographs. Figure 4.18 shows the results of the agglomerative clustering. Table 4.7 and Figure 4.19 to Figure 4.26 provide an example of a randomly selected photograph set per fine-scale group.

4.3.4.1.1 Fynbos with sandstone-quartzitic derived soils (referred to as sandstone-quartzitic fynbos from here onwards)

- Group 1 (**high-altitude fynbos**) included rocky peak high-altitude fynbos sites situated in Western Altimontane Sandstone Fynbos at a mean altitude of 2061 m (range 2057 – 2077 m) with sandstone-quartzitic derived soils and geology consisting of sandstone, minor grit, conglomerate and shale. Mean slope was 13 degrees.
- Group 2 (**sand slope fynbos**) were rocky slope or foot slope fynbos vegetation sites in Winterhoek Sandstone Fynbos situated on sandstone-quartzitic derived soils with

sandstone, minor grit, and conglomerate and shale geology. Mean altitude was 1301 m (range 661 – 1630 m) and mean slope was 13.9 degrees (range 8 – 25 degrees).

- Group 3 (**hydromorphic fynbos**) comprised hydromorphic fynbos vegetation sites in Winterhoek Sandstone Fynbos with sandstone-quartzitic derived soils on a mean slope of 3 degrees (range 0.3 – 8 degrees) and at an average altitude of 1062 m (950 – 1156 m). The geology consisted of either sandstone, minor grit, conglomerate and shale or quartzitic sandstone with thin shale and conglomerate lenses.

4.3.4.1.2 Outcrop thicket-forest (referred to as outcrop thicket-forest from here onwards)

- Group 4 (**outcrop thicket-forest**) were thicket-forest vegetation sites within Winterhoek Sandstone Fynbos but situated on rocky outcrops, with a mean slope of 20 degrees (range 7 – 27 degrees) and mean altitude of 960 m (range 899 – 1150 m). Underlying geology comprised sandstone, minor grit, conglomerate and shale and soils were partly shale- or sandstone-quartzitic derived.

4.3.4.1.3 Fynbos with shale-derived or partly shale-derived soils ***(referred to as shale fynbos from here onwards)***

- Group 5 (**shale - sandstone, grit, conglomerate**) were fynbos vegetation sites situated on partly shale-derived soils with sandstone, minor grit, conglomerate and shale geology within Winterhoek Sandstone Fynbos and comprised valley bottomland areas at a mean altitude of 1040 m (range 899 – 1141 m) and mean slope of 3 degrees (range 1 – 6 degrees).
- Group 6 (**shale slope fynbos**) included rocky or foot slope fynbos vegetation sites situated on partly shale-derived soils with sandstone, minor grit, conglomerate and shale geology and a mean altitude of 873 m (range 837 – 929 m) and mean slope 14 degrees (range 4 – 23).
- Group 7 (**shale-band fynbos**) contained fynbos vegetation sites situated on shale-derived soils within Northern Inland Shale-Band vegetation and comprised underlying geology of quartzitic sandstone with thin shale and conglomerate lenses. These two photograph sites were located on the shale-band with altitudes of 994 and 1144 m and mean slope of 5 degrees.
- Group 8 (**shale - sand, silt and mudstone**) were fynbos vegetation sites in Winterhoek Sandstone Fynbos situated on partly shale-derived soils with thinly bedded sandstone, siltstone and mudstone geology comprising a mixture of rocky slopes and seasonally wet and valley bottomland areas. The mean altitude was 1193 m (range 1089 – 1405 m) and mean slope was 12 degrees (range 5 – 22 degrees).

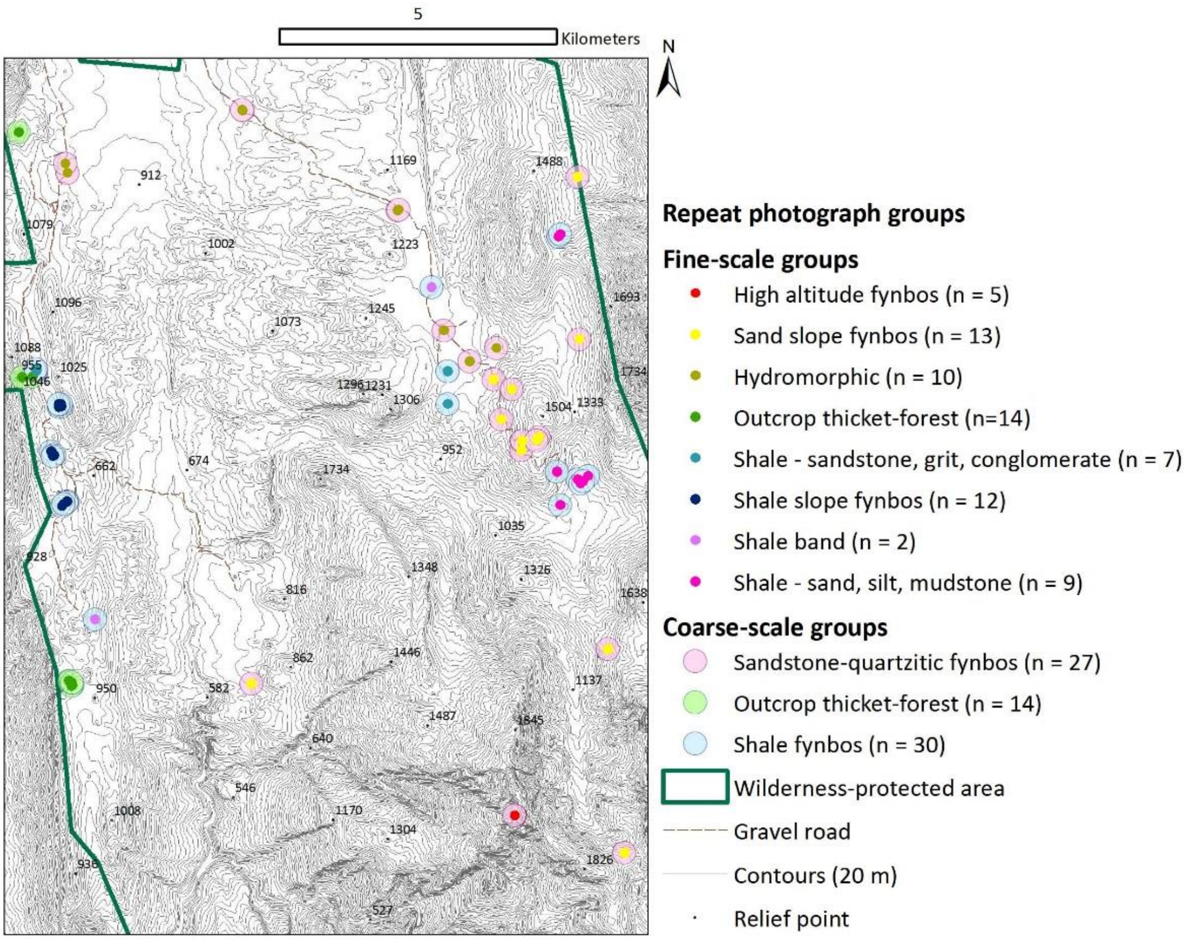

Figure 4.17 Spatial distribution of groupings of repeat photographs identified in agglomerative clustering results described above and presented in Figure 4.18 below.

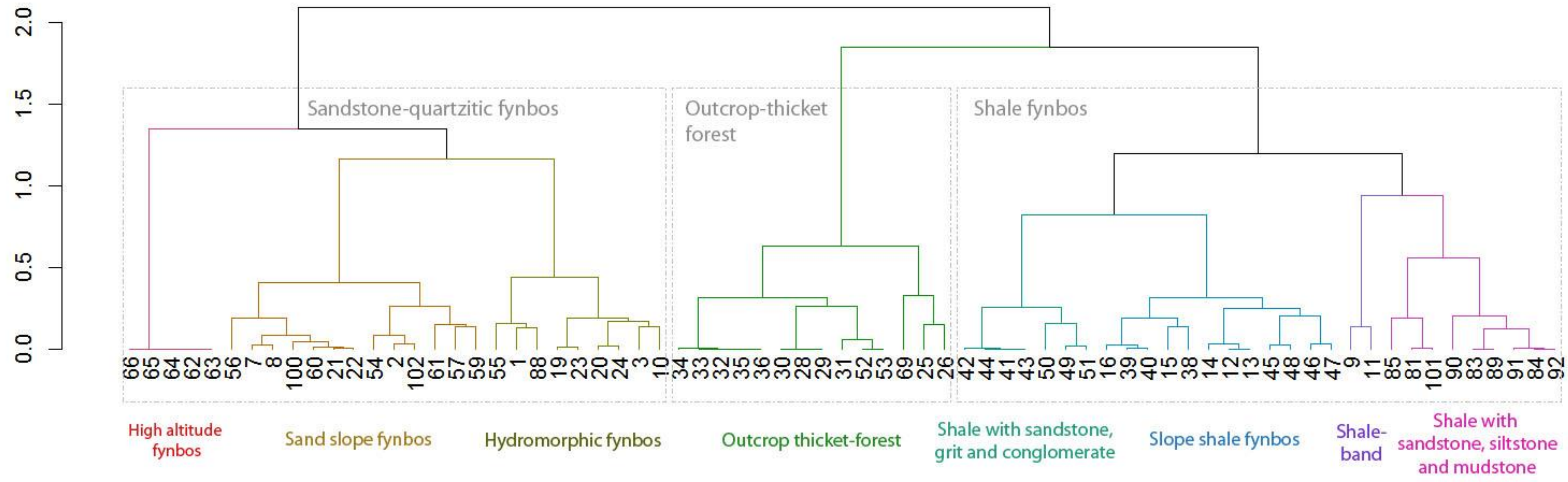

Figure 4.18 Biophysical site (dis)similarities between 71 repeat photographs taken in the Groot Winterhoek wilderness-protected area in the Cape Floristic Region Mountains. Groupings are based on agglomerative hierarchical clustering using Gower's distance for biophysical site characteristics: aspect, slope, altitude, soils, geology, local vegetation and unique site characteristics. Three coarse-scale and eight fine-scale groupings are evident. A randomly selected historical and repeat photograph set is shown in Table 4.7, Figure 4.19 to Figure 4.26.

Table 4.7 Examples of historical and repeat photographs for each agglomerative cluster type identified in Figure 4.18. Qualitative descriptions of change are provided per photograph set.

Figure 4.19 High altitude fynbos (group 1: n = 5 photos). Randomly selected image set: An increase in the height of the cover of restioids (*Restio siekerii*) is evident and there is the appearance of two shrub species (*Hebenstretia robusta* and *Erica* sp.) in the image, which were not evident in 1977. These changes have resulted in a decrease in the exposure of rock. Original photograph: FJ Kruger; Repeat: S Jack.

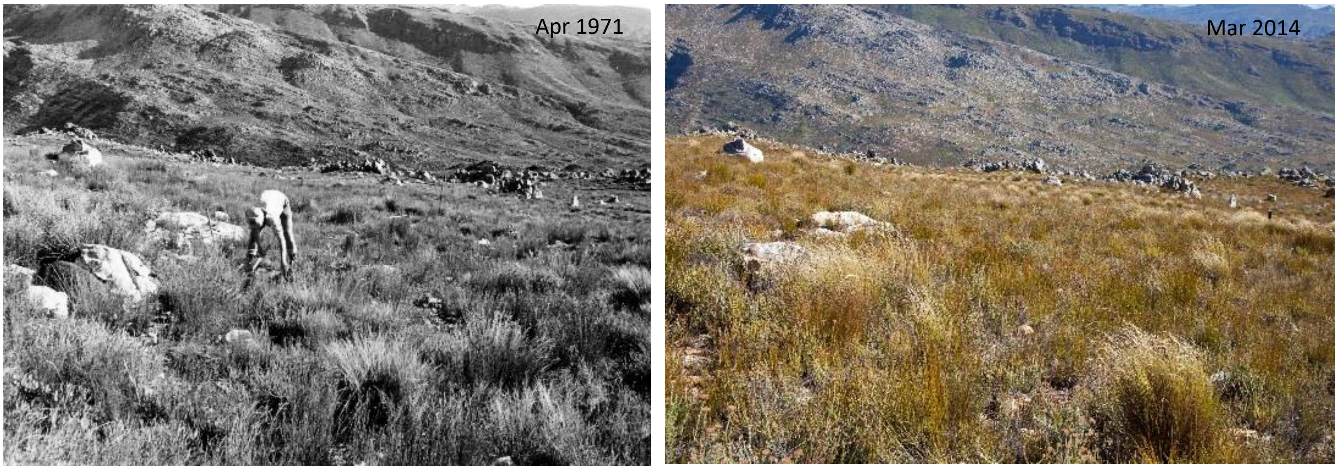

Figure 4.20 Sand slope fynbos (group 2: n = 13 photos). Randomly selected image set: A slight decline in the exposure of rock is evident because of a marginal increase across growth forms including restioids, non-proteoid shrubs (*Elytropappus glandulosus* and *Metalasia muricata*) and grasses. Original photograph: H Howe; Repeat: S Jack.

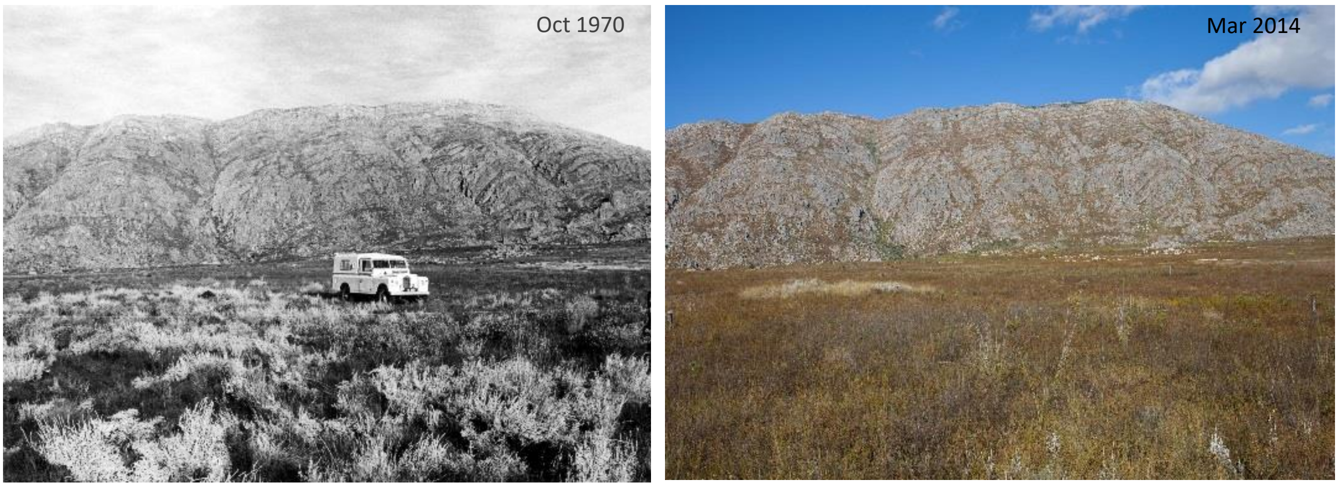

Figure 4.21 Hydromorphic fynbos (group 3: n = 10 photos). Randomly selected image set: The vegetation cover seems similar between the two images, but in terms of composition, there has been a reduction in *Stoebe plumosa* and an increase in restioids and ericoids. The *Spatella tulbaghensis* population seems unchanged. Original photograph: FJ Kruger; Repeat: S Jack.

Figure 4.22 Outcrop thicket-forest (group 4: n = 14 photos). Randomly selected image set: There has been a decline in thicket-forest canopy cover with an increase in *Stoebe* sp. along the thicket-forest margin and the loss of the large *Protea nitida* individual. Original photograph: RA Haynes; Repeat: S Jack.

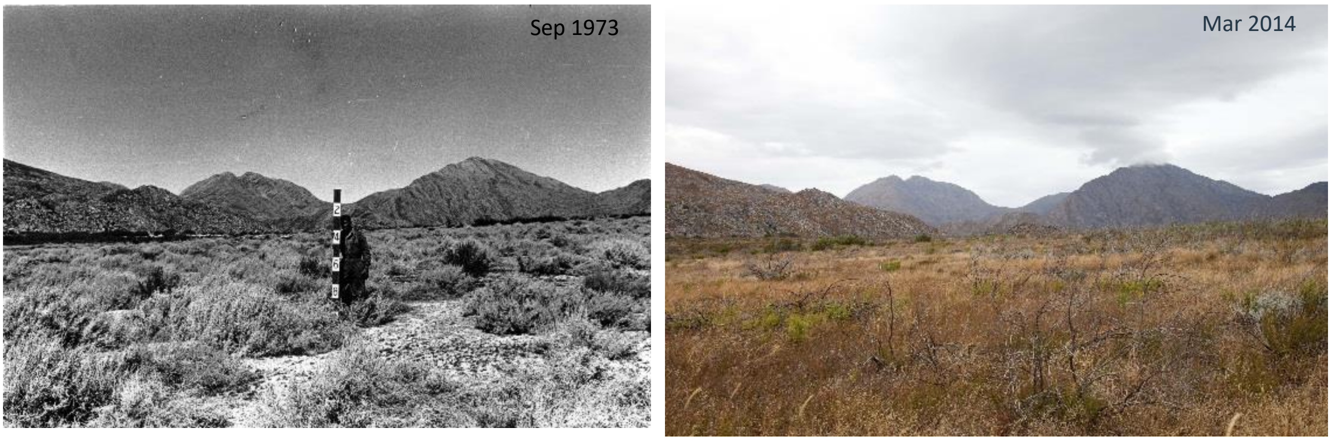

Figure 4.23 Shale - sandstone, grit, conglomerate (group 5: n = 7 photos). Randomly selected image set: A decrease in bare areas, exposed rock and *Stoebe plumosa* cover is evident. This is accompanied by an increase in restioids, grasses (*Pennisetum macrourum* and *Pentameris* sp.) and shrubs such as *Cliffortia* sp. and *Protea repens.* Original photograph: RA Haynes; Repeat: S Jack.

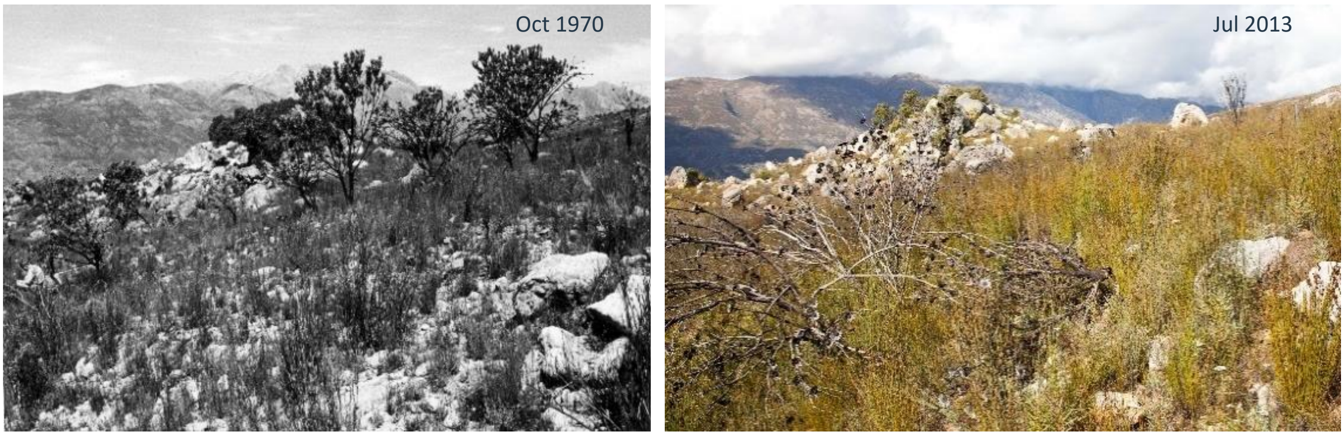

Figure 4.24 Shale slope fynbos (group 6: n=12 photos). Randomly selected image set: There has been a decrease in bare ground and exposed rock with an increase in overall vegetation cover dominated by restioids, and shrubs (*Elytropappus glandulosus* and *Protea repens* juveniles). *Protea repens* skeletons are also now present in the image and there has been a loss of the larger adult *Protea laurifolia* individuals but they are still present in the understory as seedlings. A decrease in thicket-forest vegetation is evident for the rocky outcrop in the middle distance. Original photograph: FJ Kruger; Repeat: S Jack.

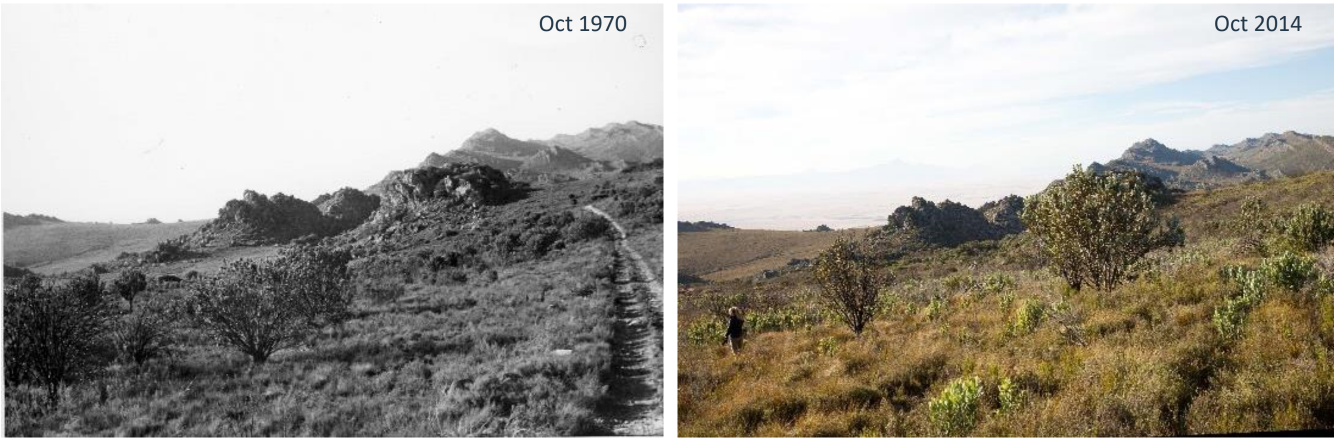

Figure 4.25 Shale-band (group 7: n = 2 photos). Randomly selected image set: A decrease in road area and a slight decline in exposed rock is evident due to an increase in vegetation cover and height. There has also been an increase in juvenile proteoids mainly *Protea laurifolia* and other shrubs including *Metalasia muricata* and *Elytropappus glandulosus*. Fewer large and taller *Protea laurifolia* individuals are now present in the image. There has been an increase in thicket-forest cover on the rocky outcrops in the distance. Original photograph: FJ Kruger; Repeat: S Jack.

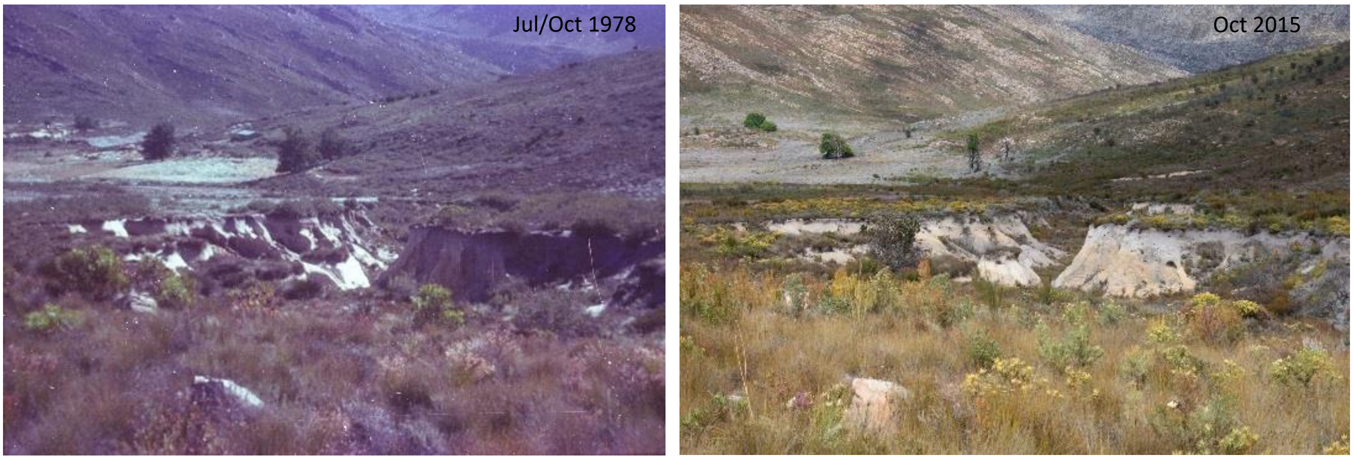

Figure 4.26 Shale - sand, silt and mudstone (group 8: n = 9 photos). Randomly selected image set: Vegetation cover has increased across the photograph except on the steep sides of the gulley where it remains similar. There has been a decline in *Stoebe* sp. cover around gully edges (particularly in the distance), which has been replaced by vegetation dominated by proteoids, including *Protea repens* and *Leucadendron* spp. including *L. rubrum and L salignum*, and ericoids. There has been a decline in *Cliffortia* sp. in the gulley and on the sides of the gulley. When considering the old cultivated fields in the left distance both *Stoebe* sp. and the cover of grasses, comprising largely *Aristida* sp., has increased. Original photograph: D Bands; Repeat: S Jack.

4.3.4.2 Change in quantitatively derived cover classes

There was an overall increase in vegetation in the repeat photographs (V = 230, p<0.0001), a decrease in bare ground (V = 895.5, p = 0.01), and the exposure of rock (V = 1617.5, p<0.0001) and no change in road cover (V = 47, p = 0.21) (Figure 4.27). When considering changes specific to coarse scale groupings, for outcrop thicket-forest sites, there was no clear trend in vegetation cover (V = 18, p = 0.22), bare ground (V = 26.5, p = 4.1), or rock (V = 83, p = 0.06) cover classes and there were no roads present in both sets of photographs. In contrast, vegetation cover increased in the repeat photographs of sandstone-quartzitic (V = 14, p<0.001) and shale (V = 41, p<0.001) fynbos clusters. For the sandstone-quartzitic fynbos the increase in vegetation cover was largely because of a decline in the exposure of rock (V = 275, p<0.001) as opposed to a decrease in bare ground (V = 50, p = 0.08) or road cover (V = 23, p = 1.05). This differed from shale fynbos where the increase in vegetation cover was largely because of a decrease in bare ground cover (V = 291.5, p<0.001) as well as a decline in exposed rock cover (V = 236, p = 0.001) with no change in road cover (V = 6, p = 0.63).

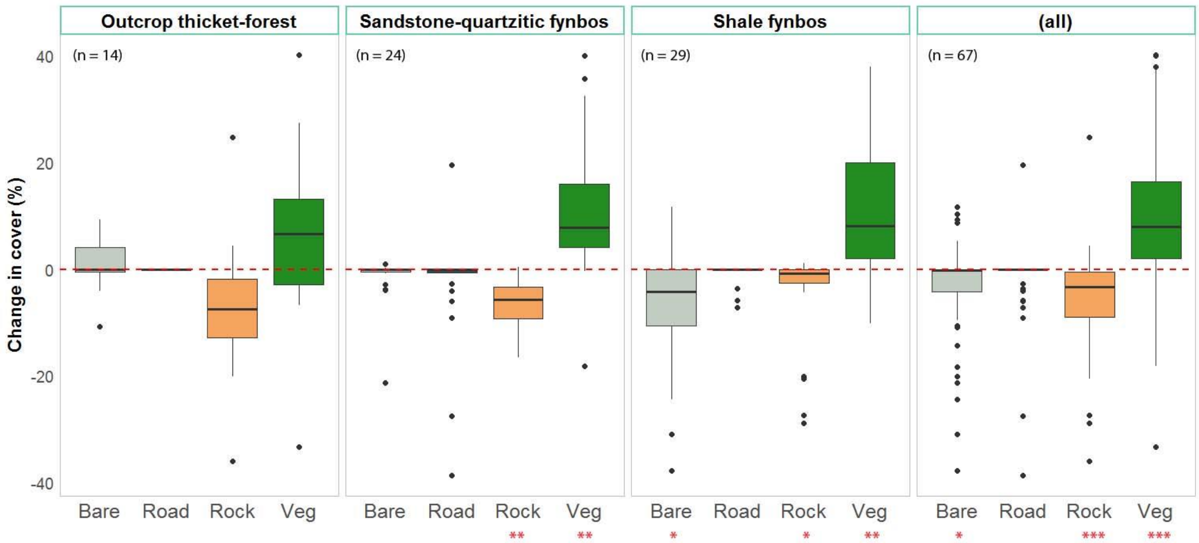

Figure 4.27 Change in the cover of bare ground, road, rock and vegetation for 67 repeat photographs differentiated into three groups defined in 3.4.1 and aggregated (all). Photographs were originally taken between 1970 and 1978 and repeated in 2013-2016. The dotted red line indicates zero change from the 1970 and 1978 photographs; positive (negative) change is above (below) the dashed red line. Asterisks indicate changes observed at p<0.01*; p<0.001**; p<0.0001***.

4.3.4.3 Change in qualitatively determined themes based on specific species and growth forms

In 92% of the images, there was an increase in cover of one or more growth forms or species (collectively termed 'cover types') while in 50% of the repeat photographs there was a decrease (Figure 4.28).

Increases in cover were recorded for non-proteoid shrubs (54% of photographs) and restioids (48% of photographs). This was followed by reseeding proteoids (38% of photographs), which comprised mostly *Protea repens* (22% of photographs), and *P. laurifolia* (19% of photographs), and resprouting proteoids (38% of photographs), namely *Protea nitida* (19% of photographs) and *Leucadendron* spp. (mainly *L. salignum*) (22% of photographs). Approximately 31% of repeat photographs were coded for an increase in *Stoebe* spp., whereby 24% showed signs of an increase in the cover of grasses. Approximately 34% of repeat photographs that contained rocky outcrop areas (n=38 photographs), showed an increase in thicket-forest sp. cover.

Decreases in cover were largely associated with *Stoebe* spp. (24% of photographs), but also included reseeding proteoids (13% of photographs) including large *P. laurifolia* (11% of photographs), and resprouting proteoids (8% of photographs), namely *P. nitida* (5% of photographs). Decreases in cover were also recorded in a limited number of photographs for grasses (4% of photographs), shrubs (2% of photographs), and restioids (2% of photographs). Approximately 21% of repeat photographs that contained rocky outcrop areas (n=38 photographs), were coded for a decrease in thicket-forest sp. cover.

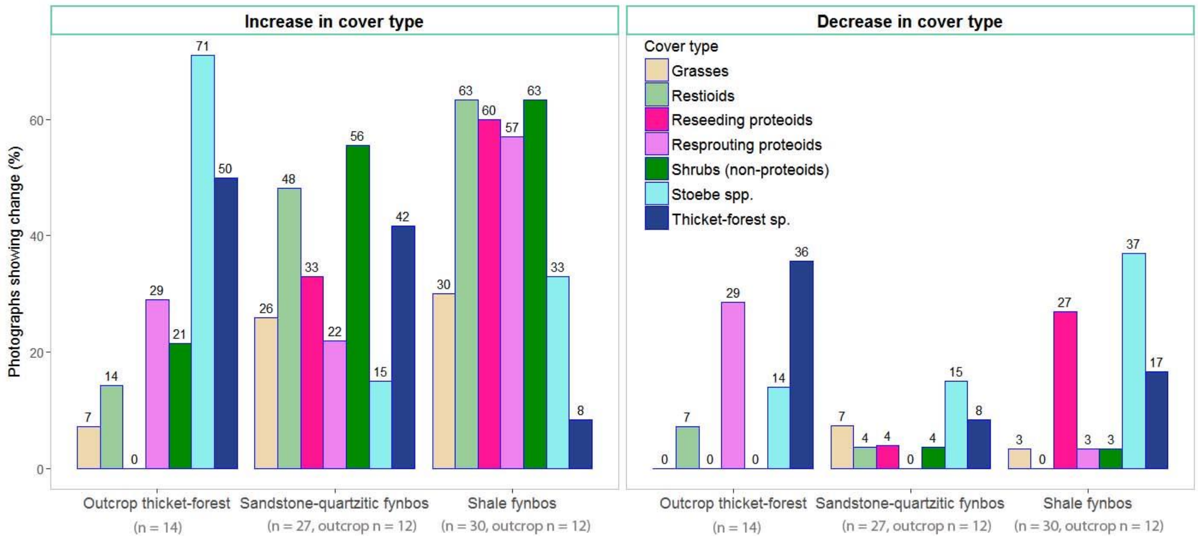

Figure 4.28 Photographs showing increases and decreases in specific growth forms or species (collectively termed 'cover types') that could be discerned in both historical and repeat photograph sets. The y-axis shows the percentage of total photographs that were coded for an increase or decrease in a specific growth form or species. Photographs are grouped according to clustering results in 3.4.1. The total number of photos visually analysed is included below the x-axis title. For thicket-forest cover types in fynbos groupings, only the number of photographs that included rocky outcrop areas were considered.

The number of photographs that showed **increases** in specific growth forms or species (referred to as cover types in Figure 4.28) differed within repeat photograph site types (*Fisher's exact p-value*: outcrop thicket-forest photographs < 0.001; sandstone-quartzitic fynbos = 0.01; shale fynbos = 0.002).

- In outcrop thicket-forest, increases in *Stoebe* spp., thicket-forest sp. and resprouting proteoids (*P. nitida*) occurred more frequently than other cover types.
- In shale fynbos, increases in shrubs, restioids and resprouting and reseeding proteoids were more frequently recorded as opposed to other cover types.
- In sandstone-quartzitic fynbos sites, increases were more frequently recorded for non-proteoid shrubs, restioids and then followed by reseeding proteoids.

- For fynbos site types, increases in resprouters comprised mainly *P. nitida* and *Leucadendron* spp. whereas increases in reseeding proteoids comprised mainly increases in young *P. repens* and *P. laurifolia*.

Decreases in cover types also differed within repeat photograph site types (*Fisher's exact p-value*: outcrop thicket-forest photographs = 0.006; sandstone-quartzitic fynbos = 0.017; shale fynbos < 0.001).

- For outcrop thicket-forest, decreases in thicket-forest sp. cover and resprouting proteoids (*P. nitida*) were more frequently recorded than other cover types.
- In contrast, decreases in *Stoebe* spp. were more frequently recorded in both fynbos repeat photograph sites as opposed to other cover types.
- For shale fynbos, decreases in reseeding proteoids were also more frequently recorded than the other cover types (Figure 4.28). Decreases in reseeding proteoids were always associated with declines in large adult *P. laurifolia* individuals that were present in historical photographs (see Figure 4.24 as an example).

4.3.5 Climate history

No annual trends were evident for total annual rainfall (Table 4.8 and Figure 4.29, A-B). However, total summer rainfall declined for both the lowland and mountain station data analysed with mean rates of change of -0.94 mm.year^{-1} and -1.88 mm.year^{-1} respectively. Although all other seasons exhibited negative rates of change for the period 1974 to 2016, they were not significant. Annual maximum and minimum mean temperatures have increased over the last 42 years (Table 4.8 and Figure 4.29, C-D). All seasons showed increasing trends except for the trend in mean winter maximum temperatures. Average seasonal rates of change for mean maximum temperatures were 0.005 (summer), 0.015 (autumn), -0.002 (winter) and 0.048 (spring) °C.year^{-1}. The trends for mean minimum temperatures were 0.001 (summer), 0.017 (autumn), 0.027 (winter), and 0.044 (spring) °C.year^{-1}. Annual mean monthly wind run declined in all seasons by -1.95 (summer), -1.58 (autumn), -1.63 (winter), -1.99 (spring) km.day.year^{-1}. There were no trends in evaporation except for annual mean spring evaporation, which showed declining trends (average rate of change -0.5 mm.year^{-1}).

Table 4.8 Trends in annual and seasonal total rainfall and mean climate data from 1974 to 2016 according to the Mann–Kendall test. The value of tau is presented which represents the direction and relative strength of the trend and p values are presented below in grey. The Theil–Sen estimator is presented in brackets, which indicates the absolute magnitude of the trend. Annual rainfall totals and climate means are aggregated for a winter rainfall year (April to March). Seasons are summer (December, January, February); autumn (March, April, May); winter (June, July, August); spring (September, October, November).

Climate variable	**Annual**	**Summer (djf)**	**Autumn (mam)**	**Winter (jja)**	**Spring (son)**
Rainfall (mm)					
Lowlands	0.03 (0.3)	-0.23 (-0.68)	-0.05 (-0.34)	0.12 (1.16)	0.02 (0.13)
p-values	*0.79*	*0.025*	*0.65*	*0.25*	*0.83*
Mountains	0.01 (0.2)	-0.28 (-1.35)	-0.04 (-0.44)	0.16 (2.3)	-0.09 (-0.7)
p-values	*0.9*	*0.009*	*0.71*	*0.14*	*0.4*
Max temp (°C)	0.51 (0.04)	0.45 (0.06)	0.48 (0.06)	0.19 (0.02)	0.29 (0.03)
p-values	*<0.00001*	*<0.0001*	*<0.00001*	*0.084*	*0.008*
Min temp (°C)	0.35 (0.03)	0.38 (0.03)	0.28 (0.03)	0.3 (0.03)	0.24 (0.02)
p-values	*<0.001*	*<0.001*	*0.01*	*0.005*	*0.024*
Wind run (km/day)	-0.66 (-1.45)	-0.66 (-1.57)	-0.62 (-1.26)	-0.61 (-1.47)	-0.6 (-1.46)
p-values	*<0.00001*	*<0.00001*	*<0.00001*	*<0.00001*	*<0.00001*
Evaporation (mm)	-0.19 (-0.25)	-0.09 (-0.26)	0.12 (0.22)	-0.19 (-0.28)	-0.29 (-0.56)
p-values	*0.07202*	*0.37*	*0.28*	*0.08*	*0.007*

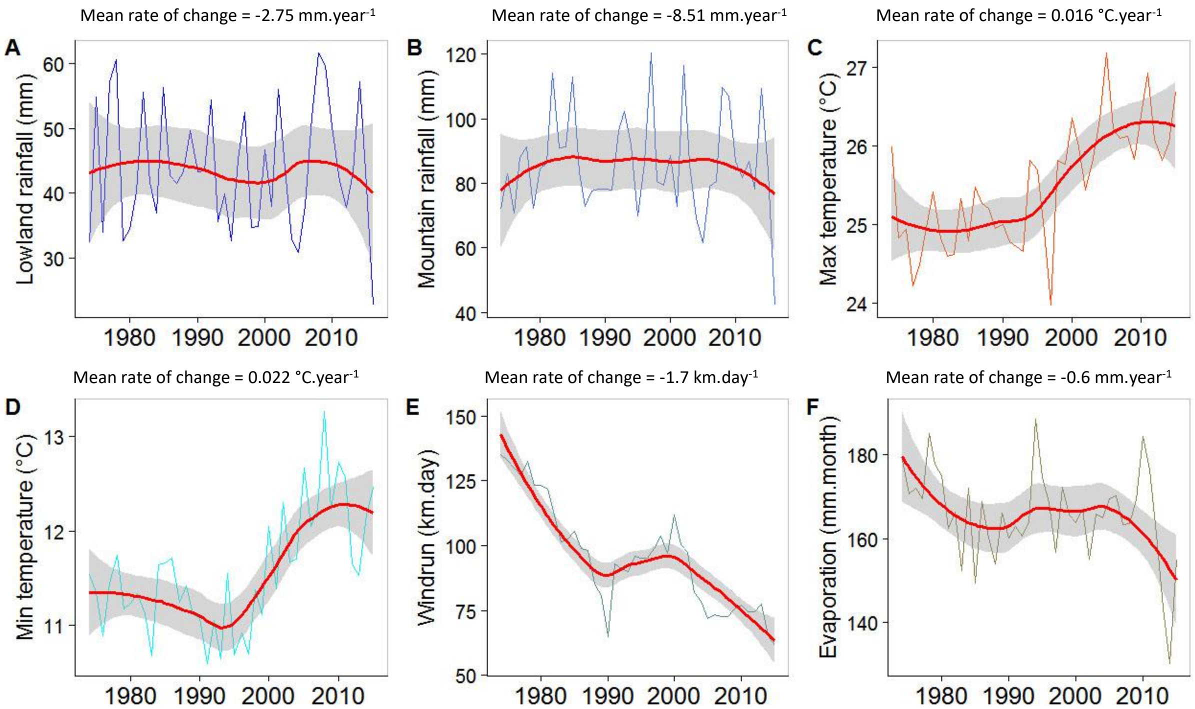

Figure 4.29 Annual trends of total rainfall data for Porterville, a lowland station (A), and a private mountain station (B) and annual trends of mean monthly climate data for Porterville (C-F). A smoothed loess curve, showing the lower and upper pointwise confidence interval around the mean, has been added to the figures for visual purposes.

4.4 Discussion

4.4.1 Changes recorded in the protected area over the last 40 years

Repeat vegetation surveys showed an increase in fynbos canopy and basal vegetation cover. In contrast, no clear trends in thicket-forest canopy and basal vegetation cover was evident for rocky outcrops. For fynbos vegetation situated on the shale-band, shifts in growth form composition were also identified. This included a switch from a historical dominance of restioids-sedges to shrubs, including non-proteoids, reseeding proteoids, e.g. *Protea repens,* and grasses, e.g. typical fynbos grass species (e.g. *Geochloa* sp.). For hydromorphic areas, on soils derived from quartzitic-sandstone, increases in vegetation cover comprised largely restioids-sedges with no shifts in other growth forms evident.

Quantitative measures of vegetation cover change from repeat photographs substantiated results from vegetation surveys. This included no change in thicket-forest vegetation cover and an increase in fynbos vegetation cover. Qualitative assessments and subsequent frequency analyses provided further depth to these results. This included evidence for an increase in pioneer species cover e.g. *Stoebe* sp*.,* especially along outcrop thicket-forest margins. For certain rocky outcrops there had also been a decline in thicket-forest canopy cover of typical fynbos thicket-forest species (such as *Heeria* spp. and *Maytenus* spp.). Increases in vegetation cover for all fynbos growth forms were recorded at sites associated with sandstone-quarzitic and partly shale-derived soils. This included an increase in non-proteoid shrubs (e.g. *Asteraceae* and *Ericaceae*), grasses, and restioids, resprouting proteoids (e.g. *Protea nitida* and *Leucadendron salignum*) and reseeding proteoids (e.g. *Protea laurifolia* and *Protea repens*). Certain photographs, however, also showed a decline in resprouting proteoids, namely *Protea nitida,* which largely occurred along outcrop thicket-forest margins. Declines were also recorded for *Protea laurifolia* especially when associated with partly shale-derived sites. The declines observed in both *P. nitida* and *P. laurifolia* comprised mainly of large individuals, at times senescent in appearance. In contrast, most increases in resprouting and reseeding proteoids were mainly related to juveniles and young plants with very large and old individuals not being observed frequently.

This is not the first study to record an increase in vegetation cover for the latter part of the 20^{th} century in the winter rainfall region of South Africa. Poulsen & Hoffman (2015) found an increase in total forest cover in the Cape Peninsula portion of the Cape Floristic Region using paired aerial and repeat ground-based photographs. Similar to changes observed in the wilderness-protected area of this study, declines in exposed rock and bare ground were observed by Poulsen & Hoffman (2015). However, in contrast to the increase in fynbos vegetation cover observed in the wilderness protected area of this study, the increase in forest observed by Poulsen & Hoffman (2015) was largely at the expense of fynbos vegetation cover. For example, only 7.5% of the repeat photographs taken by Poulsen & Hoffman (2015) showed an increase in fynbos cover. Therefore, on average, fynbos cover decreased by 5% while forest cover increased by encroaching on fynbos covered areas which were present in the original photographs. Also using repeat photographs, Hongslo et al. (2009) documented an increase in vegetation cover over the last 50 years in Namaqualand, which forms part of the Greater Cape Floristic Region (Allsopp et al. 2014) in privately owned and protected areas but not in communal lands. Working in the same region, Davis (2013; 2017) found general trends towards improved vegetation cover and composition and increased vegetation productivity for the period 1982-2011 using repeat photography and remotely sensed images respectively. Also, Hoffman & Rohde (2007) documented an increase in vegetation cover over the last half of the 20th century using repeat photographs.

To situate the influence of the wilderness-protected area into the context of global and local circumstances, two important aspects need to be addressed. The first is whether the improvement in vegetation conditions observed in this study can be linked to the establishment of the wilderness-protected area. The second is to discuss and draw conclusions about the counterfactual conditions that might have developed in the absence of the wilderness-protected area (Ferraro & Hanauer 2015; Pfaff et al. 2015; Pressey et al. 2015). This is addressed below under moderators and mechanisms of protected area impact. Lastly, a description of the potential counterfactual conditions is provided which draws on the results presented in Section 3 and Section 5 and considers land use change and drivers of change in nearby privately owned mountainous land.

4.4.2 Moderators of protected area impact: climate trends and potential effects on vegetation response

Local annual rainfall and temperature trends recorded in this study corroborate several previous analyses, which have demonstrated that the Cape Floristic Region, since the 1960s, has warmed in terms of minimum and maximum annual temperatures with no clear trends in annual rainfall (Kruger & Shongwe 2004; MacKellar et al. 2014; van Wilgen et al. 2016b). Seasonal temperature and rainfall were also similar to existing studies, except for the decline observed in summer rainfall and no trends observed for winter maximum temperatures (MacKellar et al. 2014). Declines in wind run mirror previous analyses by Hoffman et al. (2011), which showed similar results for 20 stations in the winter rainfall region of the Cape Floristic Region for the period 1974-2005. In contrast to regional findings which showed annual declines in pan evaporation over the last 30 years (Hoffman et al. 2011), only spring pan evaporation decreased in this study.

The climate trends observed in this study point towards an increase in hot, windless days, a decline in rainfall in summer months and a decline in evaporation in spring. Hot and dry conditions are particularly challenging for plants. This is because atmospheric demand for water vapor and carbon costs of maintenance respiration increase exponentially with temperature. Hot and dry conditions place both hydraulic and metabolic limitations on plants and can lead to plant death (Cramer et al. 2014). Adverse impacts of reduced rainfall have been experimentally observed for all fynbos growth form types with narrow-leaved fynbos shrubs with intermediate depth root systems being most affected (West et al. 2012). Declines in wind run can potentially amplify the negative effects of increased temperature and reduced rainfall through decreasing vegetation transpiration rates. This, in turn, can further exacerbate heat stress, especially in broad-leaved species (Hoffman et al. 2011).

Therefore, climate trends observed over the last ~40 years point towards a negative effect on plant growth and vegetation cover. However, understanding the effects of climate trends on vegetation response are complex (Hoffman et al. 2011; West et al. 2012; Altwegg et al. 2014). For example, it is possible to infer that a decline in pan evaporation in spring may represent a decline in evaporative demand during the growing season. This would mean an increase in water availability. Furthermore, decreased transpiration rates could also improve water use efficiency of photosynthesis. However, it is likely that these scenarios would have been muted by declines in summer rainfall and increases in temperature experienced in the area (Hoffman et al. 2011; West et al. 2012). Wilson et al. (2015) showed that the potential maximum biomass from the initial post fire value in the Cape Floristic Region is sensitive to temperature and precipitation extremes in the region, with less biomass accumulation in parts of the region with hot, dry summers. From this, it has been inferred that it is likely that reductions in summer rainfall and increased temperatures under future climate change would result in less biomass accumulation and therefore reduced fuel loads (Wilson et al. 2015).

Regardless of reduced summer rainfall and wind run and increased temperature trends over the last forty years an increase in vegetation cover has been observed in comparison to the baseline measures in the 1970s for the wilderness-protected area. Therefore, if the assessments of post fire maturity equivalence between historical and repeat vegetation surveys and photographs are valid, the vegetation results of this study suggest greater resilience of fynbos vegetation in the wilderness-protected area to fire events in comparison to baseline conditions. The results suggest quicker recovery of vegetation post fire regardless of the increase in temperatures and reduction in summer rainfall that has been experienced in the area. This is also despite potentially much higher intensity burns occurring (Van Wilgen 1982; van Wilgen et al. 2010).

It is likely that climate has played a moderating role and may have even reduced rates of vegetation recovery, however, it was not the dominant force shaping change in vegetation cover over the last forty years. However, it is possible that future changes in climate could play an important role in determining vegetation cover and composition and in particular, through its influence on fire patterns in the region (Altwegg et al. 2014). Furthermore, what remains uncertain is the influence of observed reductions in summer rainfall and increased temperatures on post-fire weather and the effect of this change in post-fire weather on species richness and diversity. For example, increases in cover could be dominated by species adapted to hotter and drier conditions at the expense of other less tolerant species susceptible to extreme heat or drought during post-fire conditions (Slingsby et al. 2017).

Although switches from restioids-sedges to shrubs over the last 37 years were observed for certain sites in the wilderness-protected area in this study, increases in restioids-sedges occurred in hydromorphic vegetation sampling sites. Increases in resprouting proteoids together with restioids-sedges were also recorded in sandstone and shale repeat photograph clusters. It should be noted, however, that the observed increase in mean maximum temperatures at this wilderness-protected area was half the magnitude of the increase observed by Slingsby et al. (2017). Therefore, it is likely that selections for species adapted for hotter and drier conditions may not yet be apparent for this portion of the Cape Floristic Region. Furthermore, the literature offers contradictory examples on whether resprouters or reseeders are more resilient to drought stress (West et al. 2012; Zeppel et al. 2015; Slingsby et al. 2017). Species level data, however, was not recorded during the 1970s sampling and therefore it is not clear as to whether species with higher maximum temperature tolerances dominate the increases in cover observed in this study.

4.4.3 Mechanisms of protected area impact: land use and fire, and effects on vegetation response

Vegetation in the Cape Floristic Region is strongly influenced by fire. Changes to the fire regime, e.g. fire frequency, seasonality, timing, intensity and size of fires, of an area can have important ecological implications (van Wilgen et al. 2016a). These include consequences for overall biomass accumulation each year after fire as well as species-specific responses, resulting in changes to species diversity and composition as well as overall vegetation cover (Van Wilgen 1982; van Wilgen et al. 2010).

It is clear from the literature that fire return intervals in mountain protected areas have declined since the 1970s with an increase in suitable fire-climate conditions and possibly an increase in ignition sources (van Wilgen et al. 2010). However, the importance of human ignitions prior to the 1970s have not been fully considered in these analyses in terms of changes for the 20th century. For example, using a hierarchical Bayesian model, Wilson et al. (2015) estimated that the fire return intervals for the Cape Floristic Region have shortened by approximately 4 years throughout the region when comparing 1951-1975 and 1976-2000 (Wilson et al. 2010). However, this was based on the climate and fire data for mountain protected areas for the period 1970-2000 and therefore does not reflect the potential role that humans played in the past on mountain land that was privately owned.

Grazing and burning on the mountains in the past was viewed as extremely serious in terms of the effect on the biodiversity of the region but more importantly the effects on water supplies (Drought Investigation Commission 1923; Marloth 1924; Pillans 1924; Wicht 1943; Ross & Tempel 1961; Le Roux 1966; Ackerman 1976; Bands 1977; Pooley 2012, 2015; van Wilgen et al. 2016a). The results of this study are an indication of the validity of anecdotal accounts and documentary evidence of the intensive use of fire for grazing and other purposes in the mountains of the Cape Floristic Region. Results from this study, which show changes in vegetation cover and composition, support the impression that burning was far more frequent prior to the 1970s in comparison to current burning regimes for this mountain catchment. Although the effect of the patch burning system on vegetation has never been studied, the effect of short rotation burning has been well documented for fynbos (Van Wilgen 1982). If the vegetation in the past was indeed being burnt at short rotation intervals prior to the 1970s, one would expect fire intensities to be lower than if the area was being burnt in wildfires (van Wilgen et al. 2016a).

Given the climate trends shown in this study, and observed increases in vegetation cover, it can be assumed that the wilderness-protected area currently burns less frequently than in the past. However, because of the accumulation in biomass fires are also relatively more intense than they were in the past (Wilson et al. 2015; van Wilgen et al. 2016a). This is in line with existing fire research in the mountainous uplands of the Cape Floristic Region (van Wilgen et al. 2010, 2016a). The greater intensity of fires currently being experienced in the wilderness-protected area in comparison to those of the past is also reflected through reductions in the frequency of large proteoids (Van Wilgen 1982; Vlok & Yeaton 2000; Kraaij & Van Wilgen 2014), and potentially also the increases in *Stoebe* spp. around thicket-forest margins. It is likely that low intensity fires of the past did not provide the opportunity for fire recruiting species to colonize thicket-forest margins in rocky outcrops. Declines in large proteoids included both *P. nitida*, a resprouter, but also *P. laurifolia.* The latter species is a reseeder that is known to be able to withstand cool fires when the shrub is large (Protea Atlas Project 2008). There were also declines in thicket-forest canopy cover for certain rocky outcrops, which is linked to high intensity fires (Bond et al. 2003). However, declines in thicket-forest patches were not consistently observed and there seems little evidence to suggest that the current fire regimes are causing widespread contraction of forest patches in the wilderness-protected area. This is similar to results for thicket-forest patches in the southern portion of the Cape Floristic Region (Forsyth & Van Wilgen 2008; Poulsen & Hoffman 2015).

Trends observed for vegetation on the shale-band are typical of a longer fire return interval and an increase in fire intensity in comparison to fire regimes prior to the 1970s. For example, repeated frequent and low intensity burning can result in the loss of certain reseeding shrubs (Van Wilgen 1982; Van Wilgen et al. 1994; Vlok & Yeaton 2000; Kraaij et al. 2013b). While only a small proportion of species fall into this category, they can be the dominant component of the vegetation cover (Van Wilgen 1982; Van Wilgen & Forsyth 1992). In the past, grazing and fire concentrated largely on the shale-band and near shale-derived soils as these were the most productive areas for grazing (Bands 1985). It is likely that restioids-sedges proliferated in these areas, as many are resprouters and are favoured by frequent fires (van Wilgen et al. 1992). With the release from grazing, and cessation of frequent, low intensity fires, the vegetation recovered with an increase in shrubs, dominated by reseeding proteoids and other non-proteoid shrubs. Repeated frequent and low intensity burning is known to keep the biomass low in fynbos (Van Wilgen 1982; Van Wilgen et al. 1994; Vlok & Yeaton 2000). Therefore, the higher total canopy and basal vegetation cover of today relative to the baseline measures of the 1970s is indicative of decreased fire frequency and higher intensity fires. The role of climate change on post fire weather conditions and the influence of this on species diversity and richness in this study, however, is not certain. This is an important area for future research considering observed climate trends and climate change projections (Slingsby et al. 2017).

The vegetation age of most of the sites sampled in this study prior to the most recent fire was between 14 years and 24 years, which is ideal for fynbos sustainability. In the past, the size of the individual fires would likely have covered a relatively small area and would have occurred predominantly within specific areas of the catchment. This resulted in the availability of seed from surrounding source areas which in turn enabled the regeneration of vegetation cover and composition of specific groups (Van Wilgen 1982). Despite decades, if not centuries, of patch burning this study has shown fynbos vegetation cover to be relatively resilient to these past land use practices and has been able to recover over the last forty years in comparison with baseline measures. It is, however, evident that in certain areas that were cultivated in the past, large areas of *Stoebe* spp., interspersed with grasses, still dominate the area (see Figure 4.26). Despite the appearance of restioids and proteoids (including *Protea repens* and *Leucadendron rubrum*) in these previously cultivated areas it is evident that pioneer cover largely comprised of *Stoebe* spp. and *Aristida* sp. has been seemingly resilient to change over the last 40 years. This observation highlights the persistent nature of the impacts of cultivation as a land use practice in fynbos landscapes particularly in terms of its effect on vegetation cover and diversity.

The frequency of intense fires could well increase with likely climate change scenarios forecast for the fynbos region (Wilson et al. 2010; Altwegg et al. 2015). Although it is likely that fires were frequently applied in the past to vegetation in the mountain areas, a cause for concern would be the predicted increase in frequency of very large, intense fires and the effects on mountain vegetation as well as wildlife. This is because an increase in the frequency of very large, intense fires would reduce fire return intervals over extended areas and could lead to local extinctions (Kraaij et al. 2013a), in comparison with the smaller more focused areas used for patch burning in the past. In addition, increases in fuel loads associated with warmer winter temperatures could also result in the increase in the size of individual fires, which would further increase fire frequency within local areas (Wilson et al. 2015).

In addition, the likelihood of fires burning beyond the wilderness-protected area boundary has potentially increased because of higher fuel loads. This has social and financial implications for neighbouring landowners. There is already evidence for increased occurrence of large fires in mountain protected areas in the region over the last three decades (van Wilgen et al. 2010). However, declines in rainfall may also reduce vegetation recovery rates with an opposite or no effect on fire regimes (Wilson et al. 2015). Furthermore, the effects of climate change coupled with fire disturbance on species diversity and richness and the influence of these changes on the cover of vegetation biomass directly after fire may alter the hydrology of certain areas with consequences for ecosystem services (Slingsby et al. 2017).

4.4.4 The counterfactual conditions and protected area impact

The evidence above suggests that the increase in vegetation cover in the wilderness-protected area occurred primarily because land use practices such as burning, grazing and cultivation were prohibited in the area. Therefore, in comparison to the baseline measures, while considering moderating variables and causal mechanisms, the protected area outcome can be viewed as positive for conservation. However, in terms of the counterfactual argument regarding the impact of the wilderness protected area on vegetation cover, the question that remains is whether subsistence activities would have been sustained in the area into the 21st century if the wilderness-protected area had not been established (Ferraro & Pressey 2015).

Evidence suggests that it is unlikely that grazing and patch burning would have continued into the present day regardless of protected area establishment. Substantial declines in subsistence grazing activities have been shown for privately owned mountainous areas outside the wilderness-protected area since the 1970s (see Section 3 and 5). This decline was influenced mainly by exogenous, socio-economic drivers and through land use transition pathways associated with globalisation, economic growth and development and the expansion of the market economy. This suggests that the establishment of the wilderness-protected area had little influence on the changes observed in the environment. Vegetation cover in the wilderness-protected area would probably have increased anyway because of a decline in grazing and the use of frequent burning regardless of protected area establishment.

When considering cultivation on mountain land outside the wilderness-protected area, the number of landowners using their land for cultivation has increased significantly over the last 40 years (from 12% to 32%), even though the area of land under cultivation has remained the same (see Section 3). This suggests that the subdivision of land suitable for cultivation has increased and the intensification of cultivated areas has occurred, which is also supported by an analysis of aerial photographs which showed exponential increases in dams and buildings and linear increases in roads outside the protected area on similar terrain (see Section 5). The intensification of cultivation is usually associated with increased inputs, e.g. fertilisers and pesticides, and an increase in the number of dams and other water storage facilities on private lands (MacDonald et al. 2000). If these activities had been allowed to proliferate within the main water catchment area of the Groot Winterhoek Mountains, it is likely that water quantity and quality would have been reduced. The losses in water quantity and quality and increased fragmentation that would have occurred within the wilderness-protected area if it had not been established can be viewed as significant in terms of biodiversity conservation, and ecosystem service provision. This is especially relevant considering the role that the area plays for water supply in the Western Cape (River Health Programme 2004; Nel et al. 2013a, 2013b).

Accurate fire data for this wilderness-protected area proved one of the most challenging variables for the study. Although there is anecdotal and documentary evidence of the extent and occurrence of burning in the past, there are no accurate records of fire between 1974 and 2000 and no records at all of fire prior to 1974. Fire mapping from a large set of orthorectified aerial images for the area however confirms the patchy and extensive use of fire across the landscape prior to the establishment of the wilderness-protected area (see Section 5). This additional source of visual evidence supports the contention in this study that changes in vegetation cover observed in the wilderness-protected area were due to the demise of burning and grazing as a land use practice in the area.

4.5 Conclusion

This study has demonstrated the usefulness of impact evaluation and the value of drawing on counterfactual thinking (Ferraro & Pattanayak 2006; Ferraro & Hanauer 2015; Ferraro & Pressey 2015; Pressey et al. 2015) for understanding the full impact of protected areas in the mountainous wildlands of the Cape Floristic Region. National water source areas such as the mountain catchment in this study should receive investments to enhance and maintain the catchment to manage water supply under future climate conditions. Erosion in the wilderness-protected area is still a problem with large gully areas and high intensity fires perceived to be resulting in substantial soil and water losses. At a national level, strategic investment plans are required, and these should be integrated into strategic infrastructure development plans and the National Development Plan to support the management of existing mountain catchments, which are considered important for national and regional water supplies. The value of mountain systems in the Cape Floristic Region is critical not only for water supply but also for providing biodiversity corridors (e.g. for the movement of leopards (Martins & Martins 2006)), important bird areas and refuges for specific endemics.

Section 5. Wilderness protection influences land use and water flows, but not fire and grazing, in the mountains of a global biodiversity hotspot

5.1 Introduction

This section of my thesis contributes to the literature on protected area effectiveness by determining the impact of a protected area on water-related ecosystem services using a counterfactual approach. Several different counterfactual arguments are considered by including a mountainous area outside the protected area with similar biophysical characteristics as inside the protected area. Counterfactual conditions are operationalised for comparison by generating scenarios of land use/cover for protected and private mountain land using orthorectified historical aerial imagery from 1949 and 1972 as well as current (2014) orthoimages and information from repeat landowner interviews (from Section 3) and vegetation surveys (from Section 4). The agrohydrological model ACRU is used to model streamflow for 50 years for these scenarios. This section's objectives are to:

- determine and map change in land use/cover, including fire incidence over the last 65 years outside and inside the protected area both before and after protection using orthorectified aerial images (see Appendix 5.1);
- develop counterfactual land use/cover scenarios for inside the protected area based on the above maps of land use/cover change while also drawing on repeat social surveys and interviews (Section 3), and repeat vegetation surveys (Section 4); and
- determine the influence of the protected area on hydrological response by simulating streamflow from the current protected area and the counterfactual land use/scenarios, using the agrohydrological model, ACRU (Schulze 1995; Beven 2012), and by comparing the results to a natural scenario reflecting natural catchment conditions.

5.2 Methods

5.2.1 Study area

5.2.1.1 Selection of hydrological sub-catchments

Hydrological sub-catchments in the Groot Winterhoek Mountains (see 2.1.3) selected for study were based on Quinary sub-catchments delineated by Schulze & Horan (2010). These were obtained by sub-dividing each of South Africa's 1946 Quaternary Catchments into an upper, middle and lower relatively homogeneous area in terms of climate, soil and slope using an altitude-based methodology (Maherry et al. 2013). Sub-catchments included E10C1, E10C2, and E10C3 within E10C quaternary catchment in the Olifants River secondary catchment and G10G1, G10G2 and G10G3 within the G10G quaternary catchment and G10H1 within the G10H quaternary catchment in the Berg River secondary catchment (Figure 5.1).

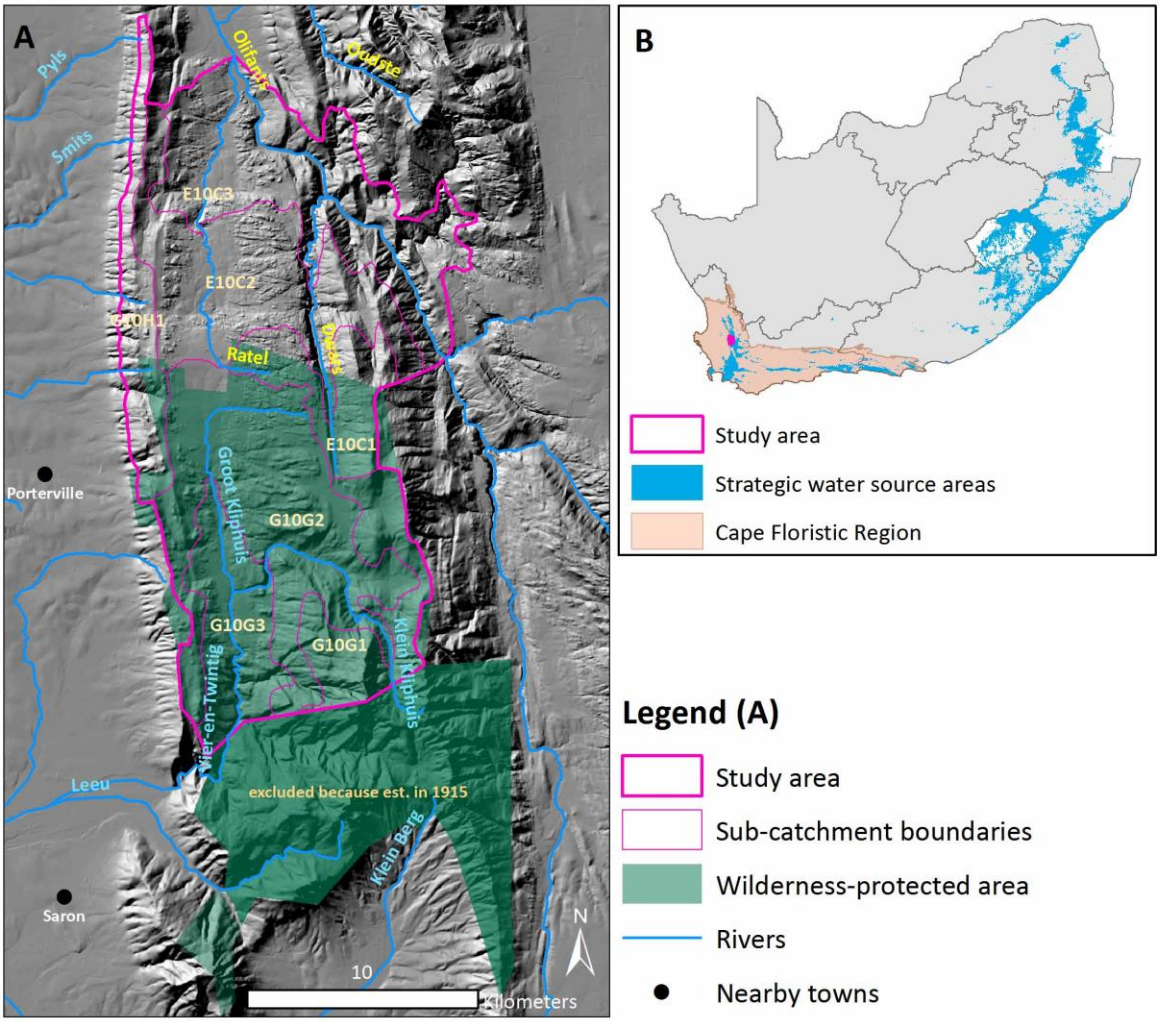

Figure 5.1 Study area: **A)** portions of the seven focal hydrological Quinary sub-catchments (E10C1, E10C2, and E10C3 for the Olifants River Catchment; and G10G1, G10G2, G10G3 and G10H1 for the Berg River Catchment) (Maherry et al. 2013) covering mountainous land outside and inside a wilderness-protected area (CSIR 2007; Schulze 2008b; SUDEM 2014; DEA 2016). **B)** The study area in relation to the Cape Floristic Region and South Africa's national strategic water source areas (SANBI 2012; CSIR 2013; MDB 2013).

Quinary sub-catchments inside the wilderness-protected area were selected to include the portion of the wilderness-protected area, which came to be protected through a series of stakeholder negotiations held over the period 1962-1978. During this process, all private activities were removed from the wilderness-protected area, with the last private landowners leaving in 1978. Therefore, the southern portion of the wilderness-protected area (~113 km^2) was excluded from the study because it has been under protection since 1915 as a state forest (Figure 5.1). The selection of Quinary sub-catchments outside the wilderness-protected area was determined by their overlap with Quinary sub-catchments within the wilderness-protected area, their location within the Groot Winterhoek Mountain landscape, as well as the availability of historical data including social survey results (Section 3) and aerial photograph coverage (Appendix 5.1).

The selected sub-catchments were clipped in ArcGIS (ESRI 2015) using the wilderness-protected area boundary to denote portions of sub-catchments outside versus inside the wilderness-protected area. This resulted in two full and three partial Quinary sub-catchments representing "outside the protected area" and covering 158 km^2 (referred to as "outside the protected area" from here onwards), and six portions of individual Quinary sub-catchments representing "inside the protected area" and covering a total of 155 km^2 (referred to as "inside the protected area" or the protected area from here onwards). Table 5.1 summarises the characteristics, determined in ArcGIS, for each of the Quinary sub-catchment portions under consideration for the study[26].

Table 5.1 Biophysical characteristics namely, slope, altitude, and mean annual rainfall for each Quinary sub-catchment portion outside and inside the protected area.

Outside/ Inside	Secondary catchment	Quinary portions	Area (ha)	Mean slope (degrees)[1]	Mean altitude (m)[1]	Mean annual rainfall[2]
Outside PA	Olifants	E10C1	846	19	1129	556
		E10C2	5903	13	808	583
		E10C3	7145	19	465	527
	Berg	G10G2	229	3	989	603
		G10H1	1756	24	763	601
Total area or overall means			***15 878***	***16***	***831***	***574***
Inside PA	Olifants	E10C1	1462	15	1180	569
		E10C2	157	21	967	573
	Berg	G10G1	2738	23	1391	589
		G10G2	7769	14	1005	605
		G10G3	2918	16	643	673
		G10H1	468	22	930	625
Total area or overall means			***15 512***	***19***	***1019***	***606***

[1](NGI-DEM 2016) *ArcGIS spatial analyst tools: surface: slope*
[2]*BioClim BIO12 Mean annual rainfall at 1 km^2 spatial resolution extracted from global set of climate layers* (BioClim 2017)

5.2.2 Mapping land use/cover change

5.2.2.1 Orthorectification of historical aerial photographs based on current orthoimages

High-resolution scans of historical aerial photograph negatives for 1949 and 1971/2 were orthorectified using a semi-automatic aerial triangulation approach with bundle block adjustment using MATCH-AT and OrthoMaster from Inpho (Trimble/Inpho 2014a) (methodology described in Appendix 5.1). A mosaic of the orthoimages was generated using automated feature-based seamlines in OrthoVista (Trimble/Inpho 2014b). For the 1971/2-

[26] ArcGIS: spatial analyst tools: zonal statistics

time step, two photograph jobs were required to obtain full coverage of the study area. These were within one year of each other and therefore the date 1972 is used to represent both sets of aerial images (Table 5.2). Colour digital orthoimages taken between December 2013 and January 2014 and with a ground sampling distance of 0.5 m were used for the orthorectification process. These current orthoimages were also used to represent the most recent time step of land use/cover. All images were obtained from the National Geospatial Information (NGI) office, Department of Rural Development and Land Reform (DRDLR), South Africa.

Table 5.2 Historical film based aerial photos including relevant scales and dates flown. Larger blocks of images were orthorectified but only the images that overlapped with the study area were used for determining changes in land use/cover, including fire (see Appendix 5.1).

Time-step	Job no.	Date flown	Scale	Area (ha)	No. of images (strips)	Focal length (mm)	Image size (cm)	Colour
1949	226	Jan-Feb 1949	1:18 000	61 000	93 (6)	152.986	23*23	greyscale
1972	676	Jan 1971	1:40 000	51 571	16 (4)	152.5	23*23	greyscale
1972	699	Feb 1972	1: 20 000	41 560	55 (5)	152.55	23*23	greyscale

5.2.2.2 Manual digitisation of buildings, cultivated areas, dams, roads and alien trees

Manual digitisation, conducted in ArcGIS (ESRI 2015), was used to delineate the boundaries of i) buildings; ii) cultivated areas; iii) water storage (dams); iv) roads; v) dense to moderate cover of alien trees (dense aliens); and vi) windbreaks and scattered alien trees (scattered aliens) in historical and current orthoimages. Only alien trees near settlements were classified. This is because alien trees in river gorges and other areas were not adequately discernible from historical greyscale orthoimages for 1949 and 1972. The total spatial coverage of each land use/cover class was determined outside and inside the protected area[27]. Rates of change were determined between the consecutive time steps i.e. i) from 1949 to 1972: 23 years apart and 6 years before protected area establishment; and ii) from 1972 to 2014: 42 years apart and 29 years after protected area establishment. The precision of manual digitisation was evaluated by re-digitising ten features in each land use class for each time step. The absolute average difference between the original measurements and re-measurements multiplied by the digitisation frequency of each land use class was used as an indication of digitisation error per time step. This is presented as error bars on the plot depicting land use change in 5.3.1.1.

5.2.2.3 Manual digitisation of fire scars

Fire scars were manually mapped in the 1949 and 1972 orthoimages in ArcGIS with support from Google Earth PRO. Super Overlays were generated for each orthoimage in Google Earth PRO. This allowed the draping of historical orthoimages over the existing Google Earth terrain

[27] ArcGIS analysis tools: statistics

model to view spatial relationships in a three-dimensional environment (Brock University 2013). Fire scars mapped within historical orthoimages were considered to represent the preceding five years of fire scars. Five years was used as a conservative estimate, as fynbos, like other Mediterranean ecosystems, requires at least 2-4 years since the previous fire in order to accumulate enough fuel to burn again (Van Wilgen 1982; Fernandes & Botelho 2003; Moritz et al. 2004; van Wilgen et al. 2010).

An inspection of the 2014 orthoimages showed that almost all the vegetation seemed to be of similar age. Therefore, digitisation of multiple individual fire scars was not possible because on initial inspection it seemed that a fire scar covered most of the study area. This assumption was supported when the CapeNature (2016b) fire database was examined. This database contains fire scars that have been accurately digitised for the study area since 2000. In the five preceding years (2009 – 2013), before the orthoimages were taken, fire had burnt 85% of the study area. In contrast, in the five years prior to 2009, i.e. from 2004 – 2008, fire had only burnt 3% of the study area. Therefore, the vegetation age was similar across 85% of the study area because of the extensive set of fires, which had occurred between 2009 and 2013. To reflect these two different fire states, two scenarios of fire scars were generated based on the CapeNature (2016b) fire database. The two fire scenarios were used to represent the two potential landscape states during the current time step. These included: i) fire scars from the five years immediately prior to when the orthoimages were taken i.e. 2009 - 2013; and ii) fire scars in the preceding five-year period i.e. 2004 – 2008. This resulted in four scenarios of fire for the study area:

- 1949, 29 years before the establishment of the protected area with fire scars likely representative of years 1944 – 1948.
- 1972, six years before the establishment of the protected area with fire scars likely representative of years 1967 – 1971.
- 2014 low burn, 36 years after the establishment of the protected area with fire scars from 2004 – 2008.
- 2014 high burn, 36 years after the establishment of the protected area with fire scars from 2009 – 2013.

5.2.2.4 Determining the spatial coverage of livestock grazing

The link between grazing and fire prior to the 1980s in the mountains of the Cape Floristic Region has been described in a number of historical accounts (Marloth 1924; Wicht 1943; Le Roux 1966; Ackerman 1972, 1976, 1979; Rabie 1974; Bands 1977), government reports (Drought Investigation Commission 1923; Ross & Tempel 1961; Ross 1963) and recent environmental history reviews (Pooley 2012, 2015; van Wilgen et al. 2016a). This has been further corroborated for the study area from interviews in 2016 with landowners surrounding the protected area (Section 3) as well as interviews carried out in 1978 with past landowners in the study area (Section 3, Bands 1985, D Bands, personal communication, March 2015).

The importance of burning to improve grazing in the past has also been described in interviews with landowners from the Cederberg region north of the study area (Bonora 2009).

Given this well-supported history of the use of fire for livestock grazing in the past, fire scars in the 1949 and 1972 orthoimages were viewed as a suitable proxy for estimating an approximate area used for grazing in 1949 and 1972. Fire is no longer used as a management tool for promoting grazing in the landscape and therefore it was not possible to use fire scars as a proxy for current day estimates. Therefore, for determining the area used for grazing in 2014, landowner survey data from 2016 within a larger portion of the privately owned catchment was used (see Section 3). This, however, was refined based on meetings with representative landowners who owned land outside the protected area. This included meeting with the chairperson and several portfolio committee members of a landowner conservancy comprising landowners that own most (73%) of the properties outside the protected area. There has been no grazing by domestic livestock inside the protected area since 1979 and therefore this was set to zero even though indigenous ungulates (e.g. grey rhebok and klipspringer) are present in the area at relatively low densities (CapeNature 2016a).

5.2.2.5 Natural vegetation

Natural vegetation was considered as being comprised of all the land that had not been delineated within the land use/cover classes described above in 5.2.2.2 – 5.2.2.4. This is because all features in the landscape, except for natural vegetation were digitised.

5.2.3 Modelling hydrological impacts of land use/cover change

There are no records of accurate streamflow for the study area. Therefore, the best method for estimating the impacts of land use/cover change on streamflow was to use a hydrological model.

5.2.3.1 Model selection

The agrohydrological model, ACRU, was used to simulate the effects of land use/cover change on streamflow outside and inside the protected area. ACRU is a multi-layer, soil-water budgeting model developed in South Africa. The model is an integrated physical conceptual model that can simulate streamflow and land use, land management and abstraction impacts on water resources at a daily time step. It is a semi-distributed model in which catchments are subdivided into hydrologically linked sub-catchments, each of which can be further sub-divided to contain one or more relatively homogenous hydrological response units, irrigated areas, dams and an area impervious to rainfall (Tarboton et al. 1992; Schulze 1995; Schulze et al. 2010a).

Although models like ACRU generally perform well at reproducing observed streamflow (Warburton et al. 2010), predictions of extreme flows are always subject to substantial uncertainties (Le Maitre et al. 2014). While the details of observed flows are unlikely to be reproduced entirely accurately, this is especially so for the Western Cape mountain catchments (New 1999, 2002). Despite shortcomings of the ACRU hydrological model, it is still well suited for the purposes of this research. This is because this study does not aim to simulate water flows accurately but rather uses the model as a comparative tool to assess the relative impacts of land use/cover change over the last 65 years in comparison with natural streamflow. ACRU has been shown to be sensitive to land use and management changes and has been used frequently in assessing these impacts on streamflow. ACRU requires relatively basic climatic data and has a large database of soils and vegetation characteristics developed specifically for South African conditions. The use of sub-catchments and hydrological response units allows for a reasonably realistic representation of spatial variations in rainfall, topography, soils, land use/cover and stream networks (Schulze 2008a). ACRU has been widely used in South Africa and elsewhere (Jewitt et al. 2004; Warburton et al. 2010, 2012; Graham 2011; Mugabe et al. 2011; Le Maitre et al. 2014; Rebelo et al. 2015; Schütte & Schulze 2017).

5.2.3.2 Model description

The ACRU model is based on a daily, multi-soil-layer water budget derived from the principles of the Soil Conservation Service Curve Number Model developed in the United States. It operates with a surface layer and two active soil horizons, the topsoil and subsoil. It is in these two soil horizons where rooting development and extraction of soil-water takes place through evaporation from the soil surface and transpiration, as well as by soil-water uptake through capillary action. Other losses occur through stormflows and saturated drainage (Schulze 1995). Figure 5.2 provides a schematic illustration of the major multi-layer, soil-water budgeting processes of the ACRU model.

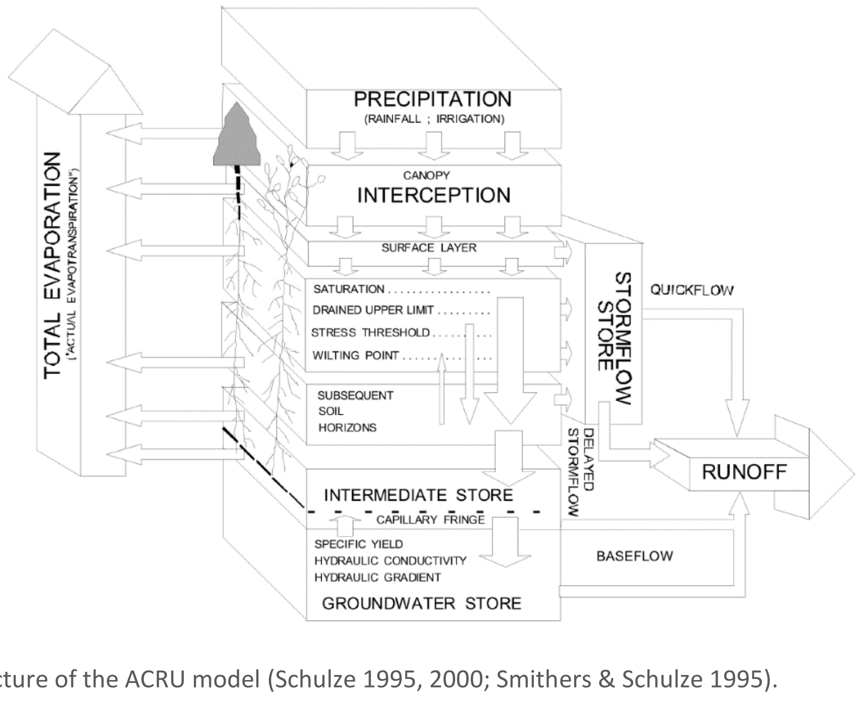

Figure 5.2 Structure of the ACRU model (Schulze 1995, 2000; Smithers & Schulze 1995).

ACRU provides output on water budget partitioning including evapotranspiration, which is partitioned into canopy intercepted losses, transpiration and soil evaporation, and streamflow which is partitioned into baseflow and stormflow. However, values for these specific components of the water budget are not available at a sub-catchment level and are only available at an individual hydrological response unit level. Furthermore, outputs for these variables at the hydrological response unit level are only available in mm. Therefore, for purposes of comparison outputs need to be converted to cubic meters for each hydrological response unit occurring within a sub-catchment. These values then need to be converted back to mm for the entire sub-catchment to determine changes in water budget partitioning at the outlet of a specific sub-catchment. Doing these conversions and the subsequent analyses required was not considered feasible within the context of this study. Therefore, this study focused on total streamflow at a sub-catchment level, but a sensitivity analyses was also conducted to understand potential changes in water budget partitioning. The modelling set up and approach used is described further below.

5.2.3.3 Model set up

In this study, the ACRU model was set up using two different approaches. This included: i) a complex scenario-based multi-sub-catchment approach to model the influence of changing land use/cover patterns on streamflow outside and inside the protected area; and ii) a simple one sub-catchment approach to test the sensitivity of streamflow to certain land use/cover types (referred to as the sensitivity analyses). The reason for these two approaches was that, as described above, extracting values for the different water budget components at a hydrological response unit level, and doing the necessary conversions for comparison and then the subsequent analyses, was not feasible. Therefore, this study focused on aggregated changes in total streamflow at the sub-catchment level for understanding the influence of land use and cover changes on hydrological response both outside and inside the protected area. The sensitivity analyses provided a way to gauge the role of specific land use/cover types on water budget partitioning and the aggregated changes in streamflow at a sub-catchment level.

5.2.3.3.1 Sub-catchment characteristics

Area, elevation and slope were determined in ArcGIS for each Quinary sub-catchment portion[28]. Daily climate for 1950-2000, including rainfall, temperature and evaporation, as well as soil characteristics for each Quinary sub-catchment portion were extracted from the Southern African Quinary Catchments Database (Schulze et al. 2010b, 2011). The Southern African Quinary Catchments Database includes unique values for all Quinary sub-catchments in South Africa for daily climate and soil properties. Methods used to generate the data in the Quinary Catchments Database are described in detail in Schulze et al. (2010b). A summary is provided below.

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[28] ArcGIS spatial analyst tools: surface: aspect and slope (ESRI 2015; NGI-DEM 2016)

- **Climate**

Daily rainfall, maximum and minimum temperature and A-pan equivalent potential evaporation values were available for the time-period 1950-2000 from the Southern African Quinary Catchments Database (Schulze & Horan 2010; Schulze et al. 2011; Maherry et al. 2013). Daily rainfall in the Quinary Catchments Database was derived by Schulze et al. (2010b) from a 50-year rainfall record extracted from a comprehensive database of daily rainfall for South Africa compiled by Lynch (2004) for a "driver" station considered most representative of the daily rainfall of each Quinary sub-catchment. Unique adjustments were made to daily rainfall values in ACRU for each Quinary sub-catchment to improve the representation of the sub-catchments areal rainfall. The month-by-month multiplication adjustment factors used in this study were developed by Schulze et al. (2010b). These adjustment factors compare the "driver" station's median monthly precipitation values to the Quinary sub-catchment's median monthly precipitation values from a spatial dataset of median monthly precipitation (~1.7 x 1.7 km resolution) developed for South Africa by Lynch (2004).

Daily minimum and maximum temperatures for the same 50-year period were derived from a spatial dataset (~1.7 x 1.7 km resolution) of daily temperatures for South Africa developed by Schulze & Maharaj (2004) for a point closest to the centroid of each Quinary sub-catchment and with altitudes most similar in terms of the Quinary sub-catchment's mean altitude. No adjustments were required in ACRU as temperature values were already adjusted for each Quinary sub-catchment to account for month-by-month temperature lapse rates based on 12 lapse rate regions identified in southern Africa (Schulze et al. 2010b). Daily estimates of solar radiation and vapour pressure deficit and from these, daily values of reference potential evaporation as well as potential crop evapotranspiration were based on those computed by Schulze et al. (2010b) using the 50 year series of daily maximum and minimum temperatures for each Quinary sub-catchment.

- **Soils**

The hydrological properties of soil in the Quinary Catchments Database were derived from those developed by Schulze & Horan (2007) using the Land Types identified in each Quinary sub-catchment on an area-proportioned basis (Schulze 2008a; Schulze et al. 2010b). These included soil-water properties, namely i) soil-water content at saturation (porosity), ii) drained upper limit (field capacity) and iii) permanent wilting point (lower limit of soil-water availability to plants) for both the topsoil and subsoil. They also included the thickness of the topsoil and subsoil and the fraction of saturated soil-water above field capacity to be redistributed daily from the topsoil to the subsoil, and from the subsoil into the intermediate groundwater store.

5.2.3.3.2 Land use/cover scenarios

Four land use/cover scenarios were developed for hydrological modelling purposes. These scenarios were based on land use/cover change mapping from orthoimages in 1949, 1972 and 2014 as described above, but also accounted for the two fire states in the current landscape setting i.e. 2014 low burn and 2014 high burn scenarios. As a result, there was one scenario each for 1949 and 1972 and two scenarios for 2014. An additional "natural" scenario was used to simulate natural streamflow conditions. This scenario represents a landscape in which no or very little human influence is evident on the mountain catchment and was used as a reference point for comparison with the other land use/cover scenarios. The five scenarios are:

1. 1949 land use and cover (Y1949)
2. 1972 land use and cover (Y1972)
3. 2014 land use and cover with small low intensity burns (2014 low burn or LB2014)
4. 2014 land use and cover with large high intensity burns (2014 high burn or HB2014)
5. Natural catchment conditions (Natural)

Fire scars for 1949 and 1972 were considered burnt and grazed in the 1949 and 1972 scenarios. In contrast, for the 2014 low and high burn scenarios, fire scars were considered representative of low and high intensity burns respectively.

5.2.3.3.3 Hydrological response units, irrigated areas, dams and impervious areas

For each land use/cover scenario, Quinary sub-catchments were subdivided into hydrological response units, irrigated areas, dams and areas impervious to rainfall. This was achieved using the land use/cover maps generated (shown in Figure 5.5 in the results) and further refined in ArcGIS through interviews (Section 3) and meetings with key representative landowners (Table 5.3). Hydrological response units in the model represented spatial segments of land for which the soil and land use/cover were assumed homogeneous in terms of the water balance and runoff generation characteristics. Irrigated areas represented different types of cultivation systems, which also differed in terms of water balance and runoff generation characteristics, but on which irrigation was applied. Dams included all artificial surface water bodies. Impervious areas included all areas impervious to rainfall from which water would flow directly onto a specified hydrological response unit.

5.2.3.3.4 Parameters for land use and cover specific variables

Parameters for land use/cover specific variables associated with hydrological response units and irrigated areas were differentiated in ACRU, guided by the assumptions set out in Table 5.3, while using the following sources:

- existing literature available for crop types in the Western Cape (Schulze 1995; Smithers & Schulze 1995; Gush & Taylor 2014);
- international guidelines for crop evapotranspiration rates (Allen et al. 1998; Van der Gulik & Nyvall 2001) but adjusted to reflect local growing, irrigation and harvesting crop cycle conditions and for evapotranspiration measured from an A-pan (Schulze 1995);
- general default parameter settings and recommendations in Schulze (1995; 1995); and
- pre-set values available in ACRU (Schulze 2008a; Warburton et al. 2012).

For hydrological response units 1-7 in Table 5.3, parameters for five land use/cover specific variables, which affect evapotranspiration from either soil or vegetated surfaces or have influences on variables affecting evapotranspiration, were differentiated and varied from month-to-month in ACRU (see Appendix 5.2, Table 3). These included: i) water use coefficients (CAY); ii) canopy interception losses per rain day (VEGINT); iii) percentage surface cover by mulch or litter (PCSUCO); iv) root mass distribution in the topsoil (ROOTA); and v) coefficients of initial abstraction (COIAM). The water use coefficient (CAY) is the proportion of water consumed by above-ground vegetation under conditions of maximum evaporation in relation to that evaporated by an A-pan in a given period. Interception per rain day (VEGINT) influences evaporation from vegetated surfaces and percentage surface cover (PCSUCO) influences evaporation from the soil surface. CAY influences evaporation from vegetated surfaces and PCSUCO influences evaporation from the soil surface. Both CAY and PCSUCO are influenced by ROOTA, which determines soil-water extraction processes by plant roots from the two soil horizons. The product of the coefficient of initial abstraction (COIAM) and soil-water content influences the rainfall abstracted by the vegetation canopy and surface litter interception, the surface detention storage and initial infiltration before stormflow commences (Schulze 1995, 2008a; Warburton et al. 2012). Therefore, the parameter set for COIAM influences the amount of rainfall that is absorbed by the soil before stormflow is generated.

The parameters used for the natural fynbos hydrological response unit (hydrological response unit 1 in Table 5.3), were based on those developed by Schulze (2004) for Acocks (1988) Veld Type 69 Macchia which covers the full extent of the study area. These were derived using a set of working rules linking parameters to climatically derived variables and physiological characteristics of the vegetation. Hydrological response units 2-4 were assumed to partition rainfall into higher proportions of stormflow and lower proportions of baseflow compared to the natural fynbos hydrological response unit. Specific characteristics included i) reduced above-ground biomass, which would reduce transpiration, vegetation interception and soil protection; and ii) reduced surface litter and mulch, which would increase soil-water evaporation and increase drying of the topsoil and decrease soil protection. In addition, certain levels of compaction of the soil surface as well as water repellences in surface and sub-surface layers were recognized and assumed to reduce infiltration rates and increase overland flow (Table 5.3). A number of pre-set parameters were available in ACRU for different levels of degraded False Macchia fynbos. These were used and adjusted slightly in this study for developing land use/cover-specific parameters for hydrological response units 2-4. False Macchia was comparable to Macchia in terms of all land use/cover-specific parameters except for VEGINT whereby Macchia had higher interception losses per rain day. The percentage of roots colonising the subsoils horizon, which affects soils water extraction (COLON), was reduced for grazing and burnt areas due to the removal of above and below ground plant material as well as poor root development in the case of grazing (Smithers & Schulze 1995).

It was estimated that 60% of the total stormflow generated would exit on the same day (Royappen et al. 2002; Warburton et al. 2012; Le Maitre et al. 2014) for all hydrological response units. It was also assumed that 0.9% of the groundwater store would become baseflow on any day. The depth of the soil from which stormflow generation occurs (SMDDEP) was set to the thickness of the topsoil for all hydrological response units except for 2, 3 and 4 for which it was decreased by 0.1, 0.15 and 0.2 m respectively (i.e. made shallower). This was to simulate more rapid saturation of the surface soils due to sub-surface water repellency or soil compaction based on assumptions set out in Table 5.3 (Schulze 2004; Le Maitre et al. 2014).

For irrigated areas, land use/cover-specific variables differentiated in ACRU included i) CAYIRR, which is the proportion of water evaporated and transpired together by the soil and vegetation under conditions of maximum evaporation in relation to that evaporated by an A-pan; as well as ii) DINTIR; and iii) COIAIR, which are analogous to VEGINT and COIAM respectively (see Appendix 5.2, Table 3 for more information). The total capacity of the dams for each sub-catchment per scenario was calculated using the all dam shapes equation specified as $V = 0.07702 * A^{1.2987}$ (Tarboton et al. 1992; Schulze et al. 2001) where V is the volume of the dam in m^3 and A is the dam surface area in m^2. Many small dams were lumped together as one dam within the model routing structure, although larger dams were kept separate. To avoid overestimating dam volume, given the non-linear form of the all shapes equation, capacity was calculated for each dam separately and then summed for dams that had been aggregated (Appendix 5.2, Table 2). Buildings, gravel and any tarred roads were represented by making that portion of the sub-catchment impervious to water infiltration. The initial amount of rainfall to be abstracted from buildings, gravel and tarred surfaces before surface runoff commences was set to 1 mm (STOIMP).

Final parameters and sources presented in Appendix 5.2, Table 3, and assumptions and sources presented in Table 5.3 were reviewed and adjusted based on ACRU-specific, expert opinion (M, Horan personal communication May to September 2017).

Table 5.3 Hydrological response units, irrigated areas, dams, and impervious areas, used in modelling streamflow outside and inside the protected area. Descriptions and assumptions, and the sources thereof, which were used to guide the parameterisation of land use/cover specific variables for each hydrological response unit and irrigated area are shown. For the specific parameters and sources used for land use/cover specific variables, see Appendix 5.2 Table 3.

Modelling units		Descriptions	Assumptions	Source of information for assumptions
Hydrological Response Units (HRU)				
1	Fynbos	• Natural vegetation mapped from 1949, 1972, and 2014 orthoimages.	Natural fynbos vegetation with no human interference	• Schulze (2004) for Acocks (1988) Veld Type 69 Macchia
2	Grazing	• Fire scars digitised from 1949 and 1972 orthoimages (see 2.2.4). • Landowner surveys (Section 3) for 2014 adjusted based on interviews with key representative landowners outside the protected area.	Fynbos 1-4 years after repeated low intensity fires but with less cover than would be expected due to fire only to account for the added impact caused by the grazing and trampling effects of livestock. Vegetation is characterised by relatively low vegetation cover (approximately reduced by 30%) dominated by resprouting restioids and sedges with relatively low abundance of tall deep-rooted shrubs (e.g. Proteaceae), and reduced presence of low shrubs of Asteraceae and Ericaceae. Soils are exposed due to trampling by livestock showing compaction and exhibit water repellences in surface and sub-surface layers.	• Information and results from i) interviews with landowners adjacent or proximal to the wilderness-protected area (**Section 3**). • Vegetation change data for the wilderness-protected area (**Section 4**). • Findings from past studies that investigated the influence of fire on vegetation and soils in the Cape Floristic Region (Van Wilgen 1981; Van Wyk 1982; Van Wilgen & Kruger 1985; Bosch et al. 1986; Higgins et al. 1987; Lindley et al. 1988; Scott & Van Wyk 1992; Scott 1993, 1997; Vlok & Yeaton 2000; Le Maitre et al. 2014; Viglizzo et al. 2016). • Bands (1985).
3	Low intensity burn fynbos	• 2004 – 2008 fire scars from CapeNature (2016b)'s fire database	Fynbos 1-4 years after small-scale fires following approximately 12-20 years of vegetation accumulation. There has been removal of some of the vegetation cover (approximately reduced by <25%) but there are substantial levels of recovery of resprouting individuals of deep-rooted shrubs (e.g. Proteaceae), as well as of restioids, sedges and low growing shrubs of Asteraceae and Ericaceae.	• Findings from past studies that investigated the influence of fire on vegetation and soils in the Cape Floristic Region (Van Wilgen 1981; Van Wyk 1982; Van Wilgen & Kruger 1985; Bosch et al. 1986; Higgins et al. 1987; Lindley et al. 1988; Scott & Van Wyk 1992; Scott 1993, 1997; Vlok & Yeaton 2000; Le Maitre et al. 2014; Viglizzo et al. 2016). • Interviews with landowner conservancy members comprising landowners that own most of the properties outside the protected area
4	High intensity burn fynbos	• 2009 – 2013 fire scars from CapeNature (2016b)'s fire database	Fynbos 1-4 years after large scale fires occurring approximately 12-20 years after fynbos vegetation accumulation. Vegetation cover has been mostly removed and reduced to sparse areas of recovering restioids and sedges (approximately reduced by 50%). There are very limited numbers of deep-rooted shrubs (e.g. Proteaceae), and low growing shrubs of Asteraceae and Ericaceae are not present. Soils exhibit high levels of water repellences in surface and sub-surface layers.	
5	Dense aliens	• Dense to moderate aliens digitised in 1949, 1972 and 2014 orthoimages.	Dense pine infestations or unmanaged plantations near settlements and farming activities.	• Interviews with landowner conservancy members comprising landowners that own most of the properties outside the protected area • Bands (1985).

6	Scattered aliens	• Land use/cover mapping of windbreaks and scattered alien trees in 1949, 1972 and 2014 from orthoimages.	Windbreaks and scattered trees comprising mostly *Acacia* tree species near settlements and farming activities.	
7	Dryland farming	• Cultivated areas digitised in 1949, 1972 and 2014 orthoimages. • Further refined into dryland farming based on landowner surveys (Section 3) and adjusted based on interviews with key representative landowners outside the protected area, and information from Bands (1985).	Mainly rooibos (*Aspalathus linearis*) farming, which is a bushy fynbos legume with needle like leaves in the family Fabaceae. Planted in winter once every five to seven years with annual harvesting occurring once a year during summer (mainly from January to March). Fields are tilled in order that rooibos seedlings can root during initial planting.	
Irrigated areas				
1	Blue berries	• Cultivated areas digitised in 1949, 1972 and 2014 orthoimages. • Further refined into specific irrigated areas based on landowner surveys (Section 3) and adjusted based on interviews with key representative landowners outside the protected area, and information from Bands (1985).	Growing season is summer with irrigation applied from spring to the end of summer (September to February) using approximately 6000 m^3 per annum per hectare. Drip irrigation is applied at a very shallow depth but frequently i.e. irrigate to 300 to 400 mm approximately 18 times a day for 5-7 minutes at a time. Plants are pruned once a year at the end of Summer whereby 30% of the plant is removed.	• Landowner interviews with landowner conservancy members comprising landowners that own most of the properties outside the protected area. • Bands (1985).
2	Apples and peaches		Growing season is summer and includes stone and pome deciduous fruit mainly apples and peaches. Orchards start losing their leaves in Autumn. In Winter, they are completely leafless. Micro spray irrigation or overhead sprinklers (depending on the land use/cover scenario) used to irrigate orchards every day during the growing season.	
3	Proteas		Plants are harvested annually from June to December. Harvesting only entails removing specific branches and therefore does not have a large impact on the overall size of the plant. Drip irrigation is applied from November to June and ranges between 3500-4000m^3 per hectare per annum. Drip irrigation or overhead sprinklers (depending on scenario) is used and the plants are irrigated approximately once a week. At times, irrigation is restricted depending on water availability.	
4	Olives		Hardier varieties such as Mission and Frantoio are grown. Micro spray irrigation is used currently. In the past overhead sprinklers were used.	

5	Buchu and other essential oils		Various indigenous and non-indigenous essential oils such as *Eriocephalus* sp. (e.g. *E. punctulatus* and *E. africanus*), *Agathosma betulina, Coleonema album, Salvia africana-caerulea* and a few lavender varieties. Harvesting taking placed between December and February depending on the species. All plants are irrigated using drip irrigation or overhead sprinklers (depending on the land use/cover scenario) during the dry summer months.	
6	Tobacco		A general tobacco crop with light irrigation applied during summer using overhead sprinklers.	
7	Artificial pastures		Consisting of a typical annual pasture crop irrigated using overhead sprinklers.	
8	Tree nuts		Deciduous nuts such as walnuts, hazelnuts and pecans as well as stone fruit nuts such as almonds. Irrigated using micro spray or overhead sprinklers depending on the land use/cover scenario.	
9	Annual crops: e.g. sweet potatoes and beans		A range of annual crop types (including mainly beans and sweet potatoes) are used for subsistence purposes and irrigated as necessary using overhead sprinklers.	
10	Citrus		An evergreen citrus crop irrigated throughout the year using drip irrigation or overhead sprinklers depending on the land use/cover scenario.	
Impervious areas to rainfall				
1	Disjunct impervious areas	• Buildings and roads digitised in 1949, 1972, and 2014 orthoimages.	Buildings and tarred and gravel roads impervious to rainfall. Water from these areas flows directly onto specific hydrological response units	• Landowner interviews with landowner conservancy members comprising landowners that own most of the properties outside the protected area. • Bands (1985).
Artificial water bodies				
1	Dams	• Dams digitised in 1949, 1972, and 2014 orthoimages.	All dams	• Landowner interviews with landowner conservancy members comprising landowners that own most of the properties outside the protected area. • Bands (1985).

5.2.3.4 Complex scenario-based modelling of land use/cover outside and inside the protected area

Quinary sub-catchment portions were divided into smaller sub-catchments to account for the influence of the protected area boundary on flow routing. This resulted in 12 and 9 smaller sub-catchments respectively inside and outside the protected area (see Appendix 5.2, Figure 1 for the flow routing diagram). Spatial coverages for hydrological response units, irrigated areas, dams and impervious areas were determined for these smaller sub-catchments in ArcGIS[29]. All sub-catchments were modelled to flow water within the general flow structure regardless of protected area boundary.

The five-land use/cover scenarios were modelled for the same time-period (1950-2000). The area, elevation, slope, climate and soils per sub-catchment did not differ among the five-land use/cover scenario model runs. Therefore, the only thing that differed among the five scenario model runs was land use/cover per sub-catchment i.e. the hydrological response units, irrigated areas, dams and area impervious to rainfall. This process was followed to isolate the impacts of land use/cover change on streamflow outside and inside the protected area. The results are five sets of simulated streamflow for 50 years representing flows from sub-catchments with different land use/cover spatial coverages per sub-catchment among scenarios but with the same area, elevation, slope, climate, and soils per sub-catchment among scenarios.

5.2.3.5 Sensitivity analysis modelling approach

It was not possible to extract values for the different water budget components at a sub-catchment level when sub-catchments were comprised of more than one hydrological response unit. Therefore, a sensitivity analyses allowed for an understanding of the influence of the different hydrological response unit parameterisations on the aggregated effects of land use/cover scenarios on streamflow at a sub-catchment level. The sensitivity analysis was used to disentangle underlying water budget partitioning processes potentially driving the aggregated effects of land use/cover on streamflow when modelling real catchment land-use patterns i.e. under the complex scenario-based modelling approach. A sensitivity analyses was conducted using one full Quinary sub-catchment (E10C1) i.e. disregarding protected area boundaries. The sensitivity analysis focused on hydrological response units 1-4 shown in Table 5.3. The sub-catchment was modelled fully comprising only one type of hydrological response unit with no other land use types. The focus was on hydrological response units that comprised large portions of the total areas outside or inside the protected area within specific land use/cover scenarios. This included the grazing and high intensity burn area hydrological response units. The sub-catchment was also modelled based on the natural hydrological response unit and the low intensity burn area hydrological response unit for comparison purposes. The main aim of this sensitivity analyses was to investigate the influence of these

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[29] ArcGIS Analysis tools: statistics

hydrological response units on the major water budget partitioning processes in the catchment to determine their role on baseflow, stormflow, soil evaporation, transpiration and canopy intercepted losses.

5.2.4 Data and statistical analysis

All data and statistical analyses were performed in R (R Core Team 2016).

5.2.4.1. Frequency of fire scars

To determine if the number of fire scars were equally common across fire scenarios i.e. 1949, 1972, 2014 low burn and 2014 high burn, the Pearson's Chi-square Goodness of fit test was used. If the fire scars were equally distributed across the scenarios, the expected proportions would be 0.25, 0.25, 0.25, and 0.25. Therefore, the null hypothesis for the test is: there is no significant difference between the observed and expected frequencies of fire scars across scenarios. While the alternative hypothesis is: there is a significant difference between the observed and expected frequencies of fire scars across scenarios[30].

5.2.4.2 Scenario-based hydrological impact analysis

Streamflow analyses focused on the mean accumulated streamflow (known as USFLOW in ACRU) for sub-catchments inside and outside the protected area. This entailed determining the mean daily streamflow for the 12 and 9 smaller sub-catchment portions outside and inside the protected area respectively for all land use/cover scenarios: Y1949, Y1972, LB2014, HB2014, and for the natural scenario for the period 1950 - 2000. Daily streamflow in mm as opposed to cubic metres was selected from ACRU for output. This can be converted to cubic metres by dividing by 1000 and multiplying by the average area of the sub-catchments inside or outside the protected area.

To compare the hydrological regime for land use/cover scenarios to the natural scenario, exceedance probabilities were computed[31] and flow duration curves were produced for daily streamflow for 1950 – 2000 outside and inside the protected area[32]. Flow duration curves have been used extensively to characterise streamflow distributions. They highlight the relationship between streamflow and the percentage that streamflow is exceeded (cumulative density function) and provide statistical information on streamflow variability. Flow duration curves combine the flow characteristics of a stream throughout the range of discharge without regard to the sequence of occurrence (Smakhtin 2001; Brown et al. 2005; Lane et al. 2005; Warburton et al. 2010; Kinoshita & Hogue 2015). In addition to flow duration curves, comparative summary statistics were also used which included total, mean and median daily streamflow for the period 1950 – 2000. Summary statistics were applied to specific divisions of the flow duration curves and were constructed to reflect to i) high flows,

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[30] R package: stats, functions: chiq.test
[31] R package: HydroTSM; functions: fdc (Zambrano-Bigiarini 2014)
[32] R package: ggplot2; functions: multiple (Wickham 2009; R Core Team 2016)

ii) mid-range condition flows (mid flows), iii) low flows. These divisions represented, respectively, exceedance probabilities of i) 0-10%; ii) 11-90% ; iii) >90% (Smakhtin 2001).

The Kolmogorov-Smirnov test was applied to compare modelled streamflow distributions for the different land use/cover scenarios in relation to the natural scenario of flow conditions. Bonferroni adjustments were computed to account for multiple comparisons for different parts of the flow duration curve. The Kolmogorov-Smirnov null hypothesis is that flow duration curves for land use/cover scenarios are from the same continuous distribution as the natural scenario flow conditions. The Kolmogorov-Smirnov test statistic (D) is the maximum vertical distance between the two curves evaluated[33] (Massey Jr 1951; Kinoshita & Hogue 2015). To test the influence of the protected area under dry and wet years, the percent difference in streamflow for each scenario from the natural scenario was plotted against total annual rainfall for each year for the period 1950 - 2000. Spearman rank correlation tests were used to gauge the level of relationship between dry and wet years and the difference in streamflow per land use/cover scenario[34].

5.2.4.3 Sensitivity analysis

A similar approach to the scenario-based hydrological impact analysis was used to compare the hydrological regime of all hydrological response units modelled in the sensitivity analyses. Exceedance probabilities were computed[35] and flow duration curves were produced for hydrological response units for daily streamflow for 1950-2000[36]. Comparative summary statistics included total, mean and median daily streamflow for the period 1950 – 2000. Summary statistics were applied to specific divisions of the flow duration curves and were constructed to reflect to i) high flows (0-10% exceedance probabilities), ii) mid flows (11-90% exceedance probabilities), and iii) low flows (>90% exceedance probabilities) (Smakhtin 2001). The Kolmogorov-Smirnov test was applied to compare modelled streamflow distributions for hydrological response units[37]. Bonferroni adjustments were computed to account for multiple comparisons for different parts of the flow duration curve. Bar charts were used to visualise changes in water budget partitioning processes between the hydrological response units modelled. Specifically, the following variables were extracted and plotted:

- canopy intercepted losses, transpiration and soil evaporation (which collectively comprises evapotranspiration); and
- baseflow and stormflow (which collectively comprises streamflow).

[33] R package: stats; functions: ks.test (Wang et al. 2003)
[34] R package: stats; functions: cor.test
[35] R package: HydroTSM; functions: fdc (Zambrano-Bigiarini 2014)
[36] R package: ggplot2; functions: multiple (Wickham 2009; R Core Team 2016)
[37] R package: stats; functions: ks.test (Wang et al. 2003)

5.3 Results

5.3.1 Land use/cover change – time steps 1949, 1972 and 2014

5.3.1.1 Change in land use/cover types and rates of change

Prior to protected area establishment in 1978, dams, buildings, roads and cultivation increased in area from 1949 to 1972 both outside and inside the protected area (Figure 5.3). After protected area establishment, the development of dams, buildings, roads and cultivated fields and orchards continued to increase outside the protected area while inside the protected area development declined. The rate of land use change differed before and after protected area establishment outside the protected area for all land use types. This included a substantial increase in area converted for water storage purposes (as dams) and for buildings after protected area establishment. In contrast, the rate of increase in area converted to roads and cultivated areas slowed after 1972. Grazing declined at similar rates both outside and inside the protected area from 1949 to 1972 before protected area establishment and from 1972 to 2014 after protected area establishment. The substantial declines in grazing resulted in increases in natural vegetation both outside and inside the protected area. Alien trees associated with human settlements and agriculture increased outside the protected area prior to protected area establishment but declined after protected area establishment. Although far fewer alien trees were associated with human settlements inside the protected area prior to establishment, rates of change for each time step followed similar trends to those outside the protected area.

5.3.1.2 Size and frequency of dams

There was an increase in the number of dams constructed outside the protected area from 1949 to 2014 (Figure 5.4 A). There was also an increase in the number of larger dams built outside the protected area between 1972 and 2014 (Figure 5.4 B). While the number of dams inside the protected area also increased between 1949 and 1972, no new dams were built in the protected area between 1972 and 2014. Furthermore, the majority of those that were present in 1972 have since become revegetated and no longer function as storage facilities (Figure 5.4 B).

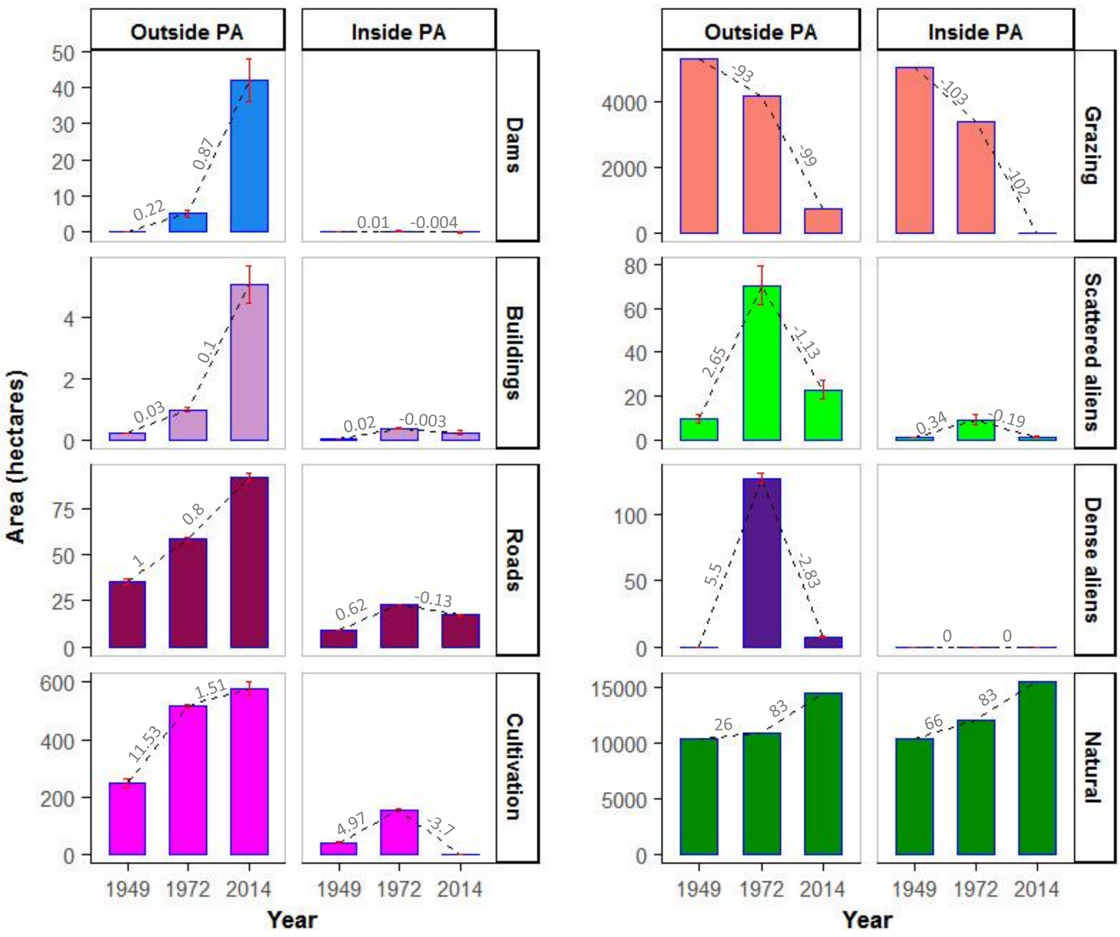

Figure 5.3 Contrasts of land use/cover change outside and inside a protected area in a mountainous catchment in the Cape Floristic Region, pre- and post-protected area establishment in 1978. The annual rate of land use/cover change (hectares.annum^{-1}) is shown in grey above the dotted line which connects successive time steps 1949, 1972 and 2014 for private land outside the protected area (15 878 ha) and currently protected land inside the protected area which was privately owned prior to 1978 (15 512 ha). Error bars are an indication of digitisation error for all land use types. This is except for grazing where the estimate is based on fire scar mapping for 1949 and 1972 and the 2014 values reflect data from land owner surveys in 2016 (see 5.2.2.3). The natural vegetation reflects area not categorised within any of the other land use/cover classes.

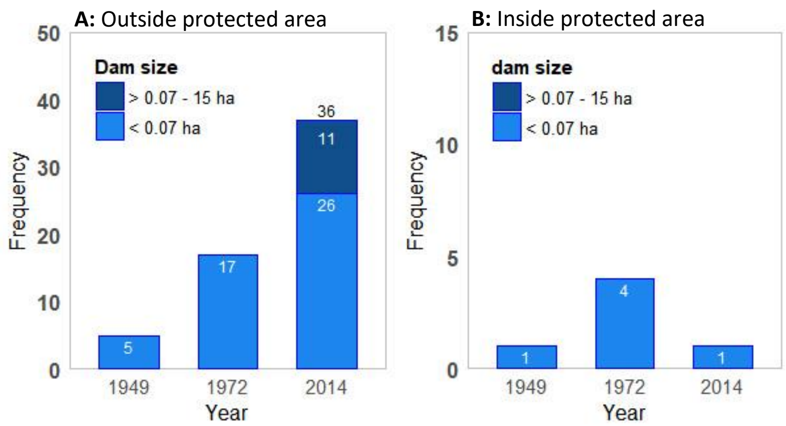

Figure 5.4 Frequency of the size of dams before and after the establishment of a protected area in 1978 for areas A) outside and B) inside the protected area.

5.3.1.3 Size and frequency of fire scars

The number of fires was more frequent in 1949 and 1972 in comparison to the 2014 low burn and high burn scenarios both outside ($\chi2$ = 327.52, df = 3, p<0.0001) and inside ($\chi2$ = 244.91, df = 3, p<0.0001) the protected area. The size of fires was also not equally distributed across the four fire scenarios. This was for both outside (total area: $\chi2$ = 13753, df = 3, p<0.0001; mean area: $\chi2$ = 8033, df = 3, p<0.0001) and inside (total area: $\chi2$ = 19140; df = 3, p<0.0001; mean area: $\chi2$ = 13415; df = 3, p<0.0001) the protected area (Table 5.4 and Figure 5.5).

Prior to protected area establishment in 1978, fire frequency was higher but the size of individual fires were smaller both outside and inside the protected area in comparison to the 2014 high burn scenario. Fire sizes were relatively small in 1949 (34-35 ha) and 1972 (30-144 ha) in comparison to the 2014 high burn scenario (3084-4771 ha). As a result, the total area burnt was greater under the 2014 high burn scenario in comparison to the 1949 and 1972 scenarios. In contrast, the total area burnt was smaller under the 2014 low burn scenario in comparison to the 1949 and 1972 scenarios. This is because, despite the frequency of fires being similar between the 2014 high and low burn scenarios, the size of fires differed.

Table 5.4 The number and size of fire scars across scenarios outside and inside the protected area (see also Figure 5.5).

	No. of fires		**Total area burnt (No. of hectares)**		**Mean area burnt (No. of hectares)**	
Fire Scenario	Outside PA	Inside PA	Outside PA	Inside PA	Outside PA	Inside PA
Pre-protected area establishment						
1949 (1944 – 1948)	154	142	5279	5016	34	35
1972 (1967 – 1971)	29	109	4179	3373	144	30
Post-protected area establishment						
2014 low burn (2004 – 2008)	3	2	271	246	90	123
2014 high burn (2009 – 2013)	4	3	12 336	14 313	3084	4771

5.3.2 Land use/cover scenarios

Land use/cover scenarios, which incorporated changes in land use and cover mapping and the four fire scenarios are shown in Figure 5.5 and a summary is presented in Table 5.5. These include:

1. 1949 land use and cover (Y1949)
2. 1972 land use and cover (Y1972)
3. 2014 land use and cover with small low intensity burns (2014 low burn or LB2014)
4. 2014 land use and cover with large high intensity burns (2014 high burn or HB2014)

The full detailed overview of land use/cover types used in hydrological modelling can be viewed in Appendix 5.2, Table 1 and Table 2. Fire scars were considered burnt and grazed in the 1949 and 1972 scenarios. In contrast, fire scars were considered representative of low and high intensity burns respectively for the 2014 low and high burn scenarios.

Table 5.5 Summary of the area of land use/cover (ha) for each land use/cover scenario. For the detailed list of irrigated areas and information on dams used for hydrological modelling, see Appendix 5.2, Table 1 and 2.

	Outside PA				Inside PA			
land use/cover	**Y1949**	**Y1972**	**LB2014**	**HB2014**	**Y1949**	**Y1972**	**LB2014**	**HB2014**
Fynbos	10 304	10 922	14 142	2575	10 444	11 952	15 246	1272
Grazing & burn	5279	4179	716	716	5017	3373	0	0
Low intensity burn area	0	0	3	0	0	0	247	0
High intensity burn area	0	0	0	11839	0	0	0	14 221
Aliens trees	10	198	31	31	1	9	1	1
Dryland cultivation	72	19	55	55	8	4	0	0
Irrigated areas	177	496	524	524	34	151	0	0
Impervious	35	59	96	96	9	23	17	17
Dams	0.3	5.3	42	42	0.07	0.25	0.06	0.06

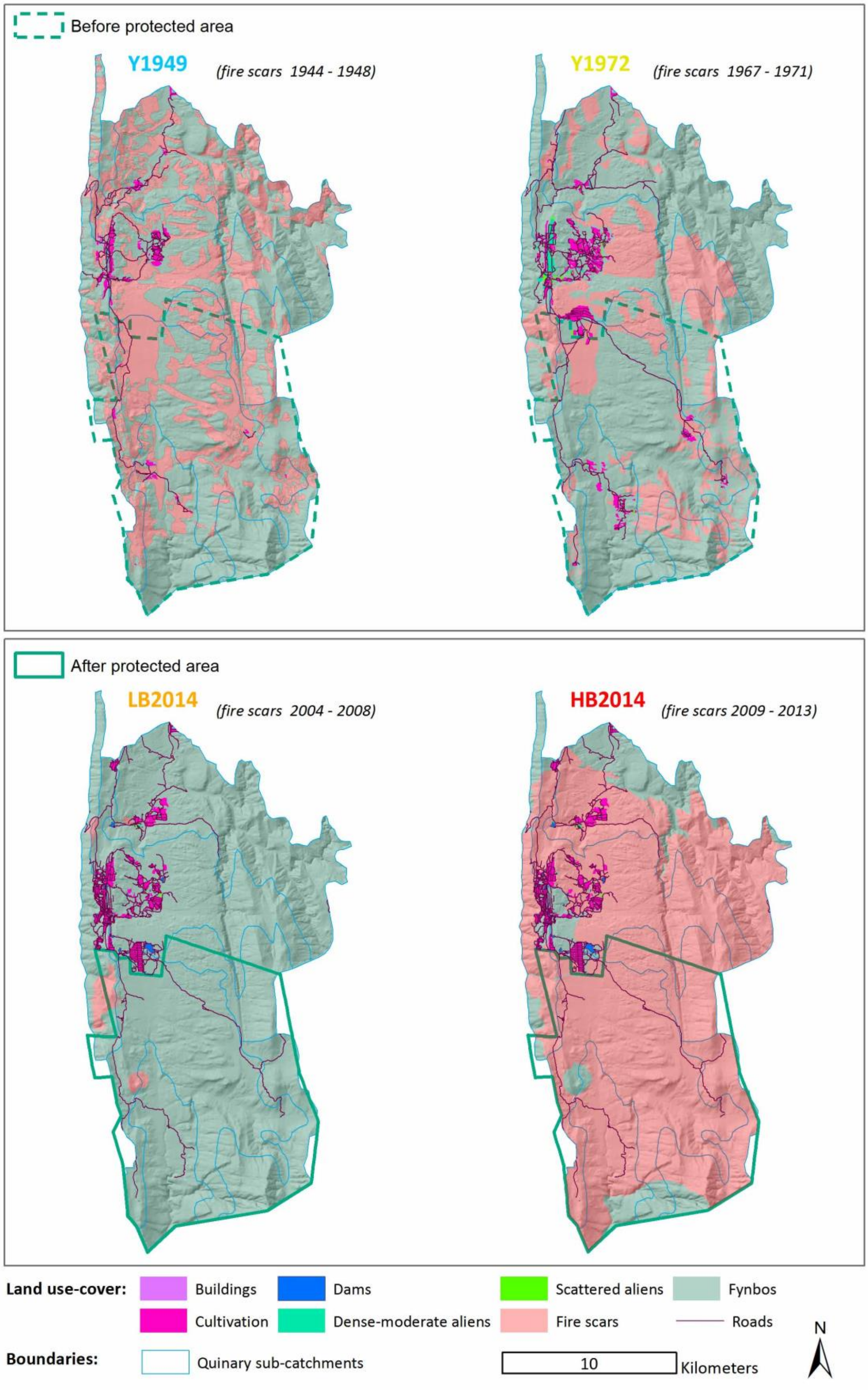

Figure 5.5 Land use/cover scenario maps, which show land use/cover and fire scars outside and inside a formal mountain protected area in the Cape Floristic Region before and after protected area establishment in 1978. Land use/cover scenarios Y1949 and Y1972 show privately owned land respectively 29 and 6 years before the establishment of the protected area ("Before protected area"). The 2014 low (LB2014) and high burn (HB2014) maps show privately owned and formally protected land 34 years after the establishment of a protected area ("After protected area"). LB2014 and HB2014 reflect two different fire scar scenarios.

5.3.3 Hydrological impacts of land use/cover scenarios

In this study, it was not feasible to extract information on water budget partitioning per hydrological response unit per sub-catchment. Therefore, a sensitivity analyses was conducted to understand the influence of the different hydrological response unit parameterisations. The sensitivity analyses provided information on the influence of different land use and cover types on the aggregated changes in streamflow resulting from real catchment land use and cover patterns captured in the land use/cover scenarios. The results of the sensitivity analyses are presented first. This is then followed by the results of the scenario-based modelling approach in which aggregated effects of different land use/cover patterns on streamflow are described for outside and inside the protected area considering each scenario separately. Lastly (in 5.3.3.3), the results from the sensitivity analyses are drawn on to understand and describe the processes driving the aggregated effects of land use patterns per land use/cover scenario on streamflow outside and inside the protected area.

5.3.3.1 Sensitivity analyses for specific hydrological response units

Flow duration curves for a sample sub-catchment modelled to contain only grazing, low intensity burn area (lowburn) or high intensity burn area (highburn) hydrological response units differed from the natural hydrological response overall and for the entire length of the flow duration curve (Figure 5.6 A and B, Table 5.6).

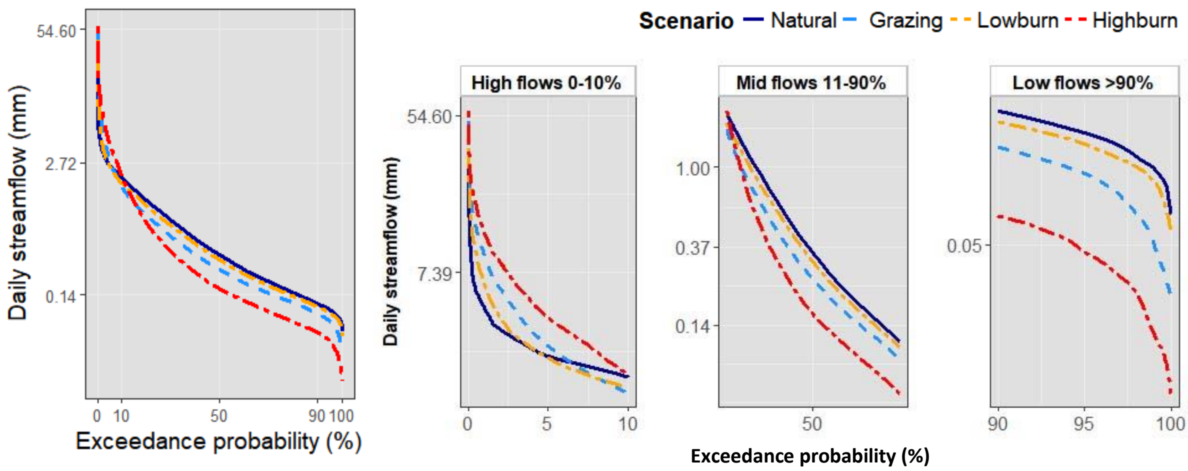

Figure 5.6 A) Daily flow duration curves for three hydrological response units from 1950-2000 in comparison to naturalised flow conditions and B) different parts of the flow duration curve showing high, mid and low flow conditions. The y-axis is on a log scale.

Changes in the water budget partitioning were largely associated with declines in canopy intercepted losses, transpiration and baseflow along with increases in soil evaporation and stormflow for the grazing, high intensity burn and low intensity burn area hydrological response units (Figure 5.7) in comparison to the natural hydrological response unit. The flow duration curves for all hydrological response units also differed from each other. The high burn hydrological response resulted in greater mean daily streamflow under high flow conditions, but less mean daily streamflow under mid and low flow conditions in comparison with the grazing and the low burn hydrological response unit. Grazing resulted in greater mean daily streamflow under high flow conditions and less mean daily streamflow under mid and low flow conditions in comparison with the low burn hydrological response unit.

The high intensity burn area hydrological response unit resulted in the greatest shift in how water was partitioned between baseflow, stormflow, soil evaporation and transpiration. This included a significant increase in stormflow and a resulting loss in baseflow. There was also an increase in water lost through soil evaporation as opposed to transpiration and canopy intercepted losses. The high intensity burn area hydrological response unit had the least amount of vegetation and litter cover in comparison to all other hydrological response units, and in comparison with the natural hydrological response unit. This was followed by the grazing hydrological response unit and then the low intensity burn area response unit. This is reflected in the parameters used for VEGINT (interception by vegetation), PCSUCO (litter cover) and COIAM (coefficient of initial abstractions) in Appendix 5.2, Table 3.

Table 5.6 Percentage difference from the natural scenario for daily stream flows for the period 1950 to 2000 for grazing, low intensity burn area (low burn) and high intensity burn area (high burn) hydrological response units in a sample sub-catchment. The Kolmogorov-Smirnov test statistic and p-value is shown and indicates the maximum vertical distance between the curves evaluated.

Daily streamflow 1950 - 2000	Natural	Grazing	Lowburn	Highburn
Total (daily mean)	13468 (0.75)	2.61	-0.59	16.64
Median flow (50%)	0.34	-27.89	-10.39	-53.35
High flow (5%)	2.54	13.73	-2.25	63.44
Low flows (95%)	0.10	-22.25	-6.72	-51.99
KS stat		*0.11*	*0.04*	*0.27*
K-S stat p-value		*<0.0001*	*<0.0001*	*<0.0001*
High flows (<=10%)				
Total (daily mean)	5630 (3.15)	43.69	13.67	95.15
KS stat		0.20	0.22	0.41
K-S stat p-value		*<0.0001*	*<0.0001*	*<0.0001*
Mid flows (11-90%)				
Total (daily mean)	7664 (0.54)	-26.94	-10.91	-39.44
KS stat		*0.14*	*0.05*	*0.34*
K-S stat p-value		*<0.0001*	*<0.0001*	*<0.0001*
Low flows (>90%)				
Total (daily mean)	174 (0.1)	-24.63	-6.9	-52.61
KS stat		*0.72*	*0.26*	*1*
K-S stat p-value		*<0.0001*	*<0.0001*	*<0.0001*

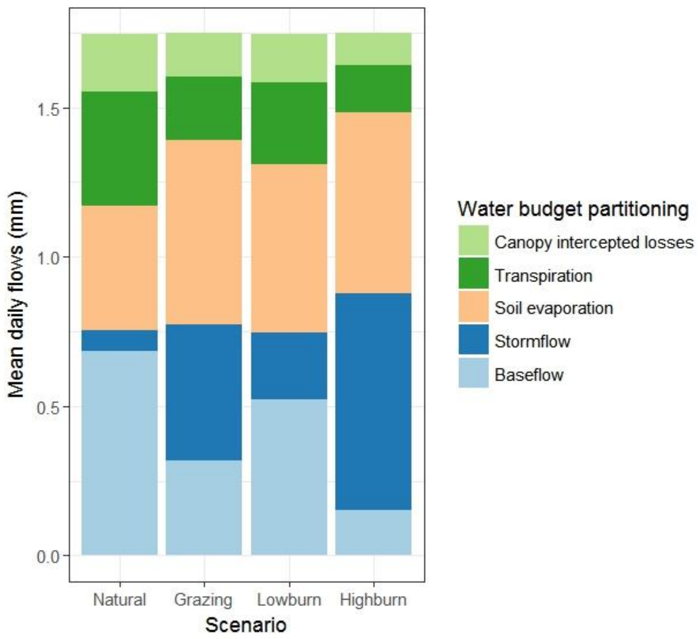

Figure 5.7 Water budget partitioning for each hydrological response unit showing increases in stormflow and soil evaporation and declines in baseflow, transpiration and canopy intercepted losses for all hydrological response units in comparison to the natural hydrological response unit.

The soil depth from which stormflow generation occurs (SMDDEP) and the coefficient of initial abstractions (COIAM) were also set at extreme limits for the high intensity burn area hydrological response unit i.e. shallower SMDDEP and lower COIAM in comparison to the natural hydrological response unit as well as in comparison to the other hydrological response units. SMDDEP and COIAM were also set lower for the grazing hydrological response unit in comparison to the natural hydrological response unit but these were not as extreme as the high intensity burn area hydrological response unit. Changes made to these parameters were set to mimic the loss of vegetation as well as increased soil-water repellences (based on assumptions set out in Table 5.3 in 5.2.3.3.4). Therefore, the extreme results modelled for streamflow under the high intensity burn hydrological response unit likely resulted from a combined influence of limited vegetation cover and litter cover and the shallower depth from which soil-water generation occurred. The lack of vegetation and litter cover would have increased soil evaporation over transpiration and canopy intercepted losses in the model. However, excessive soil drying was prevented due to the shallower depth from which soil-water generation occurred. This increased stormflow and resulted in a loss of baseflow. The results for grazing were still substantial but not as extreme due to the less extreme parameter settings used.

The low intensity burn area hydrological response unit had lower vegetation and litter cover in comparison to the natural hydrological response unit. However, the parameters set for VEGINT, PCSUCO, COIAM and SMDDEP were not as extreme as they were for the high intensity burn area or grazing hydrological response unit. Therefore, and quite intuitively, the sensitivity analyses showed that the low intensity burn area hydrological response unit resulted in a similar effect on water budget partitioning, but with reduced extremity. This included a decline in baseflow, transpiration and canopy intercepted losses and an increase in stormflow and soil evaporation. Due to the large increase in stormflow for the high intensity burn area and grazing hydrological response units there was an overall increase in daily mean streamflow (Figure 5.7 and Table 5.6). However, the loss in baseflow is also evident in the flow duration curves which showed increased streamflow under high flow conditions with subsequent large declines in mean daily streamflow under mid and low flow conditions (Figure 5.6 A and B and Table 5.6).

5.3.3.2 Scenario –based hydrological modelling results

5.3.3.2.1 Daily flow statistics and exceedance probabilities

Outside the protected area, the distribution of daily streamflow for all land use/cover scenarios differed from the natural scenario. Inside the protected area, all land use/cover scenarios except the 2014 low burn scenario differed from the natural scenario (Figure 5.8, Figure 5.9 and Table 5.7).

Outside the protected area, there was a decline in the mean daily and total streamflow for the 1972 scenario and 2014 low burn scenario relative to the natural scenario. Streamflow increased, however, in the 1949 scenario and the 2014 high burn scenario relative to the natural scenario (Table 5.7). All increases in streamflow outside the protected area were during high flow conditions (0-10% exceedance probabilities) and subsequently resulted in declines under mid to low flow conditions for all land use/cover scenarios (Figure 5.8 and Figure 5.9).

Inside the protected area, mean daily and total streamflow increased under all land use/cover scenarios relative to the natural scenario, except for the 2014 low burn scenario, which did not differ from the natural scenario (Table 5.7). Similar to the results obtained outside the protected area increases in streamflow only occurred during high flow conditions (0-10% exceedance probabilities) and declined under mid to low flow conditions (Figure 5.8 and Figure 5.9).

These results and the likely processes driving these results are described in more detail per scenario below. Annual flow duration curves showed similar trends to the constructed daily streamflow duration curves for the period 1950 – 2000 and are presented in Appendix 5.2, Figure 2 and 3.

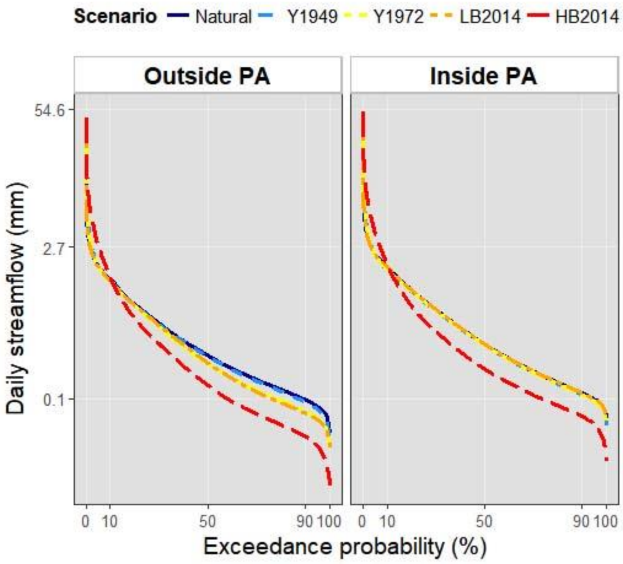

Figure 5.8 Flow duration curves for modelled daily streamflow from 1950 to 2000 for different land use/cover scenarios outside and inside the protected area compared to natural flow conditions (i.e. the natural scenario). The flow duration curves are divided into three zones of flow aligned with Figure 5.9 representing high flows, mid flows and low flows with exceedance probabilities of 0-10%, 11-90% and >90% respectively." The y-axis is on a log scale.

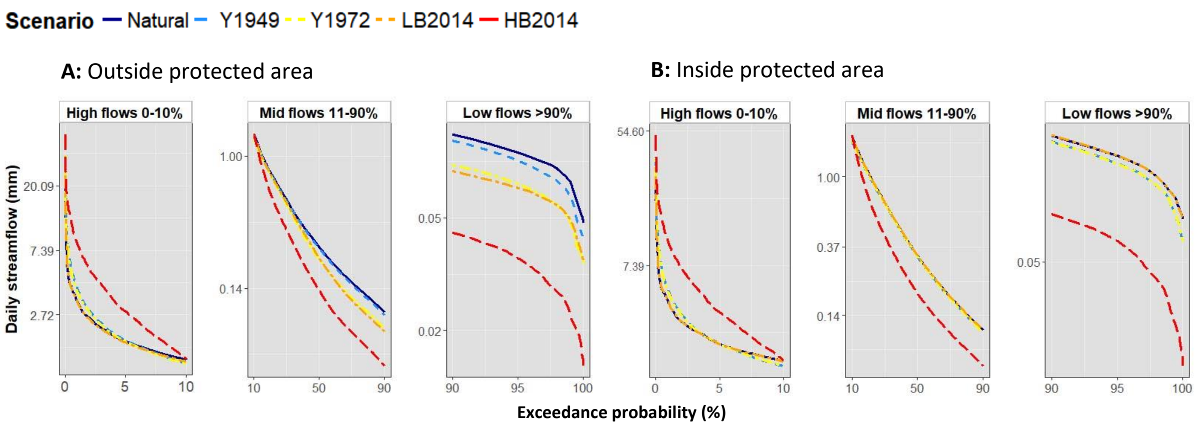

Figure 5.9 Differences in modelled daily streamflow for outside (A) and inside (B) the protected area from 1950 to 2000 for different parts of the flow duration curve and for different land use/cover scenarios (Y1949, Y1972, LB2014, HB2014), compared to the natural scenario. The flow duration curves show high flows, mid flow conditions, and low flows with exceedance probabilities of 0-10%, 11-90% and >90% respectively. These zones can be used as a general indicator of altered hydrologic conditions of the stream due to land use change. The y-axis is on a log scale.

Table 5.7 Percentage difference in daily streamflow from the natural flow conditions for 1950 to 2000 for land use/cover scenarios outside and inside the protected area. Kolmogorov-Smirnov test statistics, and p-values, are shown. This indicates the maximum vertical distance between the curves evaluated.

Daily streamflow (mm) 1950 - 2000	**Outside** the protected area (% difference to natural)					**Inside** the protected area (% difference to natural)				
	Natural	**1949**	**1972**	**LB2014**	**HB2014**	**Natural**	**1949**	**1972**	**LB2014**	**HB2014**
Total (daily mean)	9532 (0.53)	3.61	-1.97	-5.40	17.33	12344 (0.69)	-1.48	-1.45	0.03	18.20
Median flow (50%)	0.26	-1.78	-16.75	-15.56	-47.44	0.33	-1.59	-1.17	0.17	-41.50
High flow (5%)	1.80	1.22	-0.19	-1.76	58.99	2.29	1.05	0.88	-0.10	61.09
Low flows (95%)	0.08	-6.43	-22.27	-25.03	-57.68	0.10	-5.01	-5.25	0.12	-42.49
K-S stat		*0.02*	*0.11*	*0.13*	*0.32*		*0.02*	*0.02*	*0.00*	*0.22*
K-S stat p-value		*0.0008*	*<0.0001*	*<0.0001*	*<0.0001*		*0.01*	*0.009*	*1*	*<0.0001*
Extreme high flows (<=10%)										
Total (daily mean)	3954 (2.21)	11.97	8.92	-1.07	88.35	5063 (2.83)	9.08	8.06	0.36	88.56
K-S stat		*0.10*	*0.12*	*0.07*	*0.37*		*0.13*	*0.13*	*0.01*	*0.40*
K-S stat p-value		*<0.0001*	*<0.0001*	*0.004*	*<0.0001*		*<0.0001*	*<0.0001*	*0.999*	*<0.0001*
Mid flows (11-90%)										
Total (daily mean)	5431 (0.38)	-2.2	-9.3	-8.0	-32.3	7108 (0.50)	-2.89	-2.37	0.11	-30.39
K-S stat		*0.03*	*0.13*	*0.16*	*0.39*		*0.02*	*0.02*	*0.00*	*0.27*
K-S stat p-value		*0.0004*	*<0.0001*	*<0.0001*	*<0.0001*		*0.02*	*0.014*	*1*	*<0.0001*
Low flows (>90%)										
Total (daily mean)	148 (0.08)	-7.02	-23.29	-25.41	-58.46	174 (0.1)	-5.02	-5.15	0.14	-44.09
K-S stat		*0.20*	*0.75*	*0.82*	*1.00*		*0.16*	*0.17*	*0.02*	*0.99*
K-S stat p-value		*<0.0001*	*<0.0001*	*<0.0001*	*<0.0001*		*<0.0001*	*<0.0001*	*0.93*	*<0.0001*

- **2014 high burn land use/cover scenario (HB2014)**

The distribution of daily streamflow for the 2014 high burn scenario differed from natural flow conditions both inside and outside the protected area. Streamflow increased by 88% both outside and inside the protected area under high flow conditions (0-10 exceedance probabilities). In contrast, declines in streamflow occurred during mid and low flow conditions (11-90% and >90% exceedance probabilities). During mid flow conditions, the magnitude in the decline was similar outside and inside the protected area (~30 and ~32% respectively). In contrast, during low flow conditions declines in streamflow were greater outside the protected area. There was ~14% additional streamflow lost outside the protected area in comparison to inside the protected area under the 2014 high burn scenario.

- **2014 low burn land use/cover scenario (LB2014)**

The 2014 low burn scenario showed a decline in streamflow outside the protected area but had no influence on the distribution of flows inside the protected area in comparison with the natural scenario. Declines outside the protected area occurred for the whole length of the flow duration curve and ranged from ~1% under high flow conditions to ~25% under low flow conditions. Mean daily streamflow and total streamflow declined outside the protected area by ~5% in comparison with the natural scenario.

- **1972 land use/cover scenario (Y1972)**

Streamflow under the 1972 land use/cover scenario differed from natural flow conditions outside and inside the protected area. Mean daily and total streamflow was reduced outside the protected area by ~2%. The opposite occurred inside the protected area where mean daily and total streamflow increased by ~2%. However, increases only occurred under high flow conditions. These were of a similar magnitude inside and outside the protected area (i.e. ~8% and ~9% respectively). There were also decreases in streamflow under mid and low flow conditions, but these were greater outside the protected area relative to those inside the protected area.

- **1949 land use/cover scenario (Y1949)**

The 1949 scenario resulted in increases in mean daily and total streamflow both outside and inside the protected area. Similar to the high burn and 1972 scenario increases were only evident during high flow conditions with subsequent declines under mid and low flow conditions. Increases and declines were of similar magnitudes outside and inside the protected area.

5.3.3.2.2 Minimum, maximum and median flows

In terms of annual minimum, maximum and median values the results showed reductions in minimum flows and median flows outside the protected area for all scenarios but especially the 1972 scenario, and 2014 low and high burn scenarios. In contrast, there was an increase in maximum annual flow for the 2014 high burn scenario. Inside the protected area, the 2014 low burn scenario was most similar to the natural scenario in terms of annual maximums, medians and minimums. Declines were evident for annual minimums for the 1949, 1972 and 2014 high burn scenario. Similarly, to outside the protected area the 2014 high burn scenario showed increases in maximum flows inside the protected area (Figure 5.10).

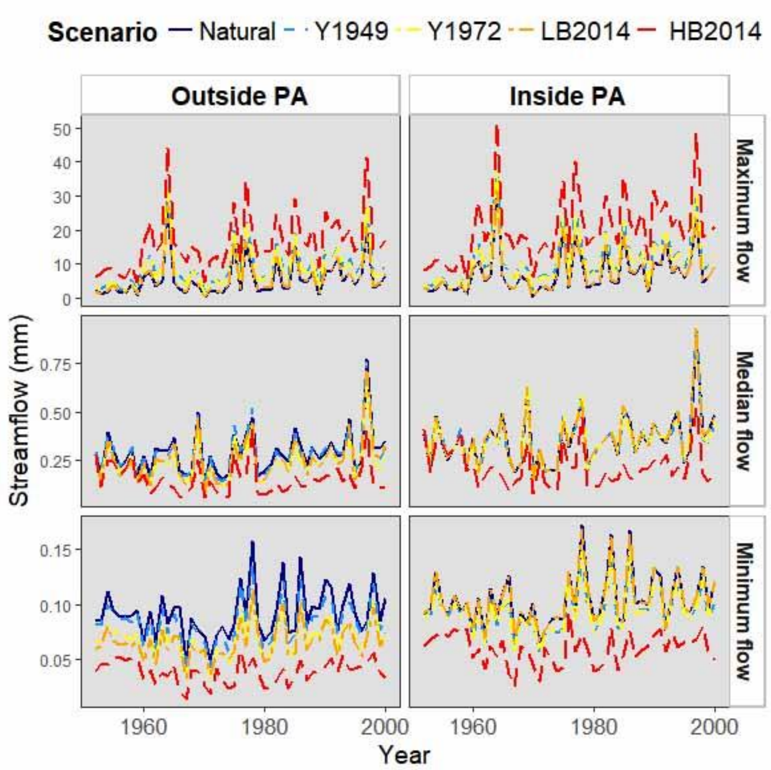

Figure 5.10 Annual maximum, minimum and median daily streamflow values for 1950 -2000 for land use scenarios in comparison to the natural scenario for sub-catchments outside and inside the protected area.

5.3.3.2.3 Protected area influence under dry and wet years

There was a positive relationship ($\rho = 0.86$, $p < 0.00001$) between annual total rainfall and the difference in annual mean daily streamflow from the natural scenario outside the protected area under the 2014 low burn scenario. Specifically, at low annual rainfall totals, there were larger percentage declines in streamflow from the natural scenario than at higher annual rainfall totals (Figure 5.11). There was no relationship for the 1972 scenario ($\rho = 0.03$, $p = 0.86$) and there were weak negative relationships for the 1949 scenario ($\rho = 0.\ 13$, $p = 0.03$) and the 2014 high burn ($\rho = -0.34$, $p = 0.02$) scenario. Inside the protected area, there were also weak negative relationships between annual total rainfall and the difference in annual mean daily streamflow from the natural scenario for all scenarios totals (Y1949: $\rho = 0.29$, $p = 0.04$; Y1972: $\rho = -0.31$, $p = 0.04$; LB2014: $\rho = -0.31$, $p = 0.03$; HB2014: $\rho = -0.39$, $p = 0.01$). See annual flow duration curves for 1950 – 2000 in Appendix 5.2, Figure 2 and 3.

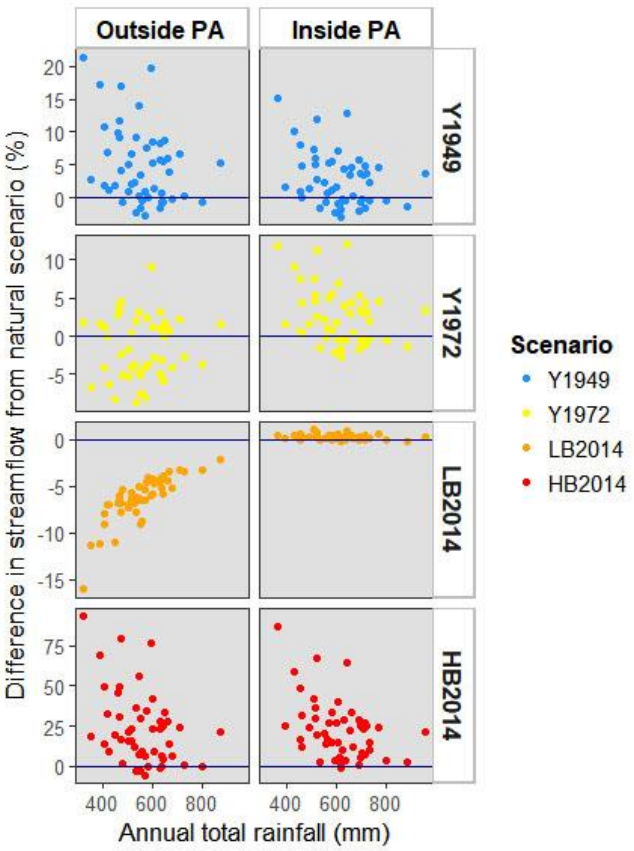

Figure 5.11 Relationship between annual total rainfall and the difference in annual mean daily streamflow from the natural scenario (indicated by the blue horizontal line at 0) for 1950 – 2000 for land use/cover scenarios outside and inside the protected area. Outside the protected area, the 2014 low burn scenario showed a strong positive relationship with annual total rainfall, showing larger declines in streamflow under dry years as opposed to wet years.

5.3.3.3 The influence of land use/cover types on streamflow in relation to the protected area

Under the 1949 scenario, there were similar trends between outside and inside the protected area in terms of streamflow deviations from the natural scenario (Table 5.8). This is because grazing comprised the dominant form of land use and cover and the area under grazing was similar outside and inside the protected area. For example, ~33 and 32% of the total area outside and inside the protected area was respectively comprised of the grazing hydrological response unit in the 1949 scenario. Therefore, the increase in streamflow during high flow (~9-12%) conditions and the decline in streamflow during mid (~2-3%) to low (~5-7%) flows in comparison to the natural scenario both outside and inside the protected area is expected (Table 5.8). In the sensitivity analyses, the grazing hydrological response increased flows by 44% under high flow conditions and reduced flows by ~24 under mid flow conditions and ~27% under low flow conditions in comparison to the natural hydrological response unit (see 5.3.3.1). These changes were largely due to the parameterisation of the grazing hydrological response unit, which had lower vegetation cover, and certain levels of soil water repellences in comparison to the natural hydrological response unit. As a result, greater water was partitioned to stormflow and soil evaporation in comparison to baseflow, transpiration and canopy intercepted losses (see sensitivity analyses results above in 5.3.3.1).

Table 5.8 Summary of Table 5.7 showing percent change in mean daily streamflow for each land use/cover in comparison to the natural scenario of streamflow outside (purple) and inside (green) the protected area. Total mean daily streamflow as well as high, mid and low flow conditions shown.

	Total		high		mid		low	
Land use/cover scenario	**Out-PA**	**In-PA**	**Out-PA**	**In-PA**	**Out-PA**	**In-PA**	**Out-PA**	**In-PA**
1949	+4	-2	+12	+9	-2	-3	-7	-5
1972	-2	-2	+9	+8	-9	-2	-23	-5
2014 low burn	-5	0	-1	0	-8	0	-25	0
2014 high burn	+17	+18	+88	+88	-32	-30	-59	-44

Similarly, to the 1949 land use/cover scenario, the 1972 land use/cover scenario mostly comprised the grazing hydrological response unit e.g., ~26 and 21% of the total area respectively outside and inside the protected area comprised the grazing hydrological response unit. Therefore, increases in high flows and declines in mid to low flows were also expected. However, increases and reductions modelled under the 1972 land use/cover scenario should have been slightly less than modelled under the 1949 land use/cover scenario due to the decline in grazing between 1949 and 1972. Inside the currently protected area, the magnitude of change in streamflow under mid and low flow conditions from the natural scenario was similar to those modelled under the 1949 scenario. However, there was a slight reduction in the increase in high flows of ~1%. It is likely that these deviations under mid and low flow conditions remained due to the increase in cultivation and dams inside the protected area. Outside the protected area, despite the increase in high flows also showing slight reductions (of ~2%), there were additional losses in streamflow under mid to low flow conditions. For example, losses in streamflow under mid to low flow conditions were greater by respectively ~7 and ~16%. These additional losses were likely due to the combined influence of the increase in alien trees, irrigated areas and dams, which added to the influence of grazing that was still dominant in the catchment.

The 2014 high burn scenario showed extreme deviations in streamflow from the natural scenario in comparison to the deviations modelled for all other land use/cover scenarios. The 2014 high burn scenario clearly represents a loss in baseflow and an increase in stormflow. For example, extreme increases (~88%) in high flows and drastic low flows (~44 and ~59%) were modelled for streamflow in comparison to the natural scenario. Under the 2014 high burn scenario ~75 and ~91% of the total area respectively outside and inside the protected area was comprised of the high intensity burn hydrological response unit. This hydrological response unit was the most extreme in terms of its influence on water budget partitioning (see results from a sensitivity analysis in 5.3.3.1). This hydrological response unit was parameterised to reflect low vegetation cover and litter cover and to mimic soil-water repellences. This resulted in most rainfall leaving the catchment via stormflow as opposed to baseflow. Therefore, the 2014 high burn scenario was mostly influenced by both the area covered and the extreme parameter settings used for the high intensity burn area hydrological response unit. The extreme nature of this scenario was evident both outside and inside the protected area. There were, however, higher losses of streamflow outside the protected area under low flow conditions in comparison to inside the protected area. This is despite a larger proportion of the sub-catchments inside the protected being comprised of the high intensity burn area hydrological response unit. Therefore, it is likely that other land use/cover classes other than the high intensity burn area hydrological response were contributing to these additional losses of streamflow outside the protected area.

The 2014 low burn scenario had the exact same spatial coverage of land use/cover classes as the 2014 high burn scenario (see Table 5.5 and Appendix 5.2, Table 1). The 2014 low burn scenario, however, did not include the high intensity burn area hydrological response unit. Instead, the 2014 low burn scenario included the low intensity burn area hydrological response unit. However, only a very small portion of the sub-catchments outside and inside the protected area was comprised of the low intensity burn area hydrological response unit. For example, only 0.02 and 1.5% of the total area respectively outside and inside the protected area was comprised of the low intensity burn area hydrological response unit. Therefore, the reductions in streamflow modelled for the 2014 low burn scenario outside the protected area were likely not due to the parameters set for the low intensity burn area hydrological response unit. Furthermore, the area inside the protected comprised higher proportions of the low intensity burn area hydrological response unit but showed no deviations from the natural scenario of streamflow. This highlights the negative influence of other land use/cover types on streamflow outside the protected area.

Grazing would have been expected to decrease streamflow under mid to low flows and increase streamflow under high flows (based on results of the sensitivity analyses in 5.3.3.1). However, the magnitude of deviations from natural streamflow should have been far less than what was modelled under the 1949 land use scenario. This is because the grazing hydrological response unit comprised a third of the area in the 1949 scenario, whereas it only comprised 4% of the area under the 2014 low burn scenario. The 2014 low burn scenario had 84% less area occupied by alien trees in comparison to the 1972 land use/cover scenario. It furthermore had almost double the impervious area to rainfall. Both impervious areas and alien trees have shown to have disproportionate effects on streamflow in comparison to the area occupied in a catchment. Impervious areas have been shown to increase streamflow under all flow conditions. In contrast, alien trees have been shown to decrease streamflow under all flow conditions (Warburton et al. 2012). Therefore, based on the decline in grazing and alien trees and the increase in impervious areas from 1972, increases in streamflow under all flow conditions would have been expected. In contrast, the 2014 low burn scenario, outside the protected area, was the only scenario that showed reductions in streamflow across the full portion of the flow duration curve.

Given that reductions were modelled under all flow conditions, it is likely that the dams and irrigated areas were influencing the hydrological response outside the protected area under the 2014 low burn scenario. There was an exponential increase in the size and capacity of the dams outside the protected area since 1972. The irrigated area also increased from 1972. Therefore, in comparison to the 1972 land use/cover scenario, it is likely that more rainwater was stored in dams. Because of this, more water was available for irrigation, which was mostly applied during the dry summer months. This would have increased losses of water through evaporative processes from irrigated areas and from dams. The avoided increase in dams and irrigated areas inside the protected area coupled with the substantial declines in grazing have thus resulted in the 2014 low burn scenario inside the protected area being very similar natural streamflow conditions. For areas outside the protected area, it is likely that any increase in flow expected from an increase in impervious areas and the decline in alien trees was dampened by the increase in dams and irrigation outside the protected area. Therefore, regardless of the declines in grazing experienced outside the protected area, the exponential increase in dams and the increase in irrigated areas reduced streamflow under all flow conditions.

5.4 Discussion

This study provides support for protecting mountainous areas that are "high and far away" (Joppa & Pfaff 2009) and which contain water source areas critical for human wellbeing, but the important question to ask is, who's wellbeing is at stake?

Findings indicate that the protected area proved significant in avoiding reductions in streamflow particularly so during mid and low flow conditions and in dry years. In the absence of the protected area, there would have been an increase in dams, buildings, roads and an intensification of cultivation in the area currently protected. This would have resulted in significant declines in streamflow. These declines were avoided by the establishment of the protected area. Protected area gains, however, were confounded by the influence of changing fire patterns on streamflow both outside and inside the protected area. For example, although there was a decline in grazing and in the incidence of frequent small-scale fires over time, this was coupled with an increase in the size and intensity of fires across the entire mountain catchment. When these changes were incorporated into the modelling process, results showed significant increases in streamflow under high flow conditions and subsequent large reductions in streamflow under mid to low flow conditions for sub-catchments both outside and inside the protected area.

The positive impact of the protected area on downstream water supply should be contextualised with the following issues in mind. There are spatial trade-offs in the benefits and costs that come from protecting the environment. In the context of this study there have been spatial trade-offs between upstream and downstream areas around the protected area. All the summer flow and >70% of the winter flows that come from inside the protected area are channelled in a concrete canal to Voelvlei which provides water to the Western Cape Water Supply System and used mostly for domestic and economic activities in urban areas. Downstream agricultural activities also benefit directly from the canal and through linkages with the Western Cape Water Supply System. Therefore, in this mountain catchment there have been trade-offs between socio-economic development which has occurred outside the protected area and the protection of water resources inside the protected area for the benefit of downstream users. Protection of water inside the protected area has come at a local cost firstly to the individuals that owned land inside the protected area before it was expropriated and secondly the lost opportunity costs related to agricultural and ecotourism activities that have developed outside the protected area.

5.4.1 Protected area impact on land use/cover and streamflow

The results of this study show that grazing intensity and fire frequency declined in this mountain catchment regardless of the protected area boundaries. Furthermore, there was an increase in fire size and intensity across the mountain catchment. Therefore, these changes cannot be ascribed to the establishment of the protected area. This contrasts with observed changes in cultivation, roads, buildings and dams, which expanded outside the protected area and declined inside the protected area after its establishment. Therefore, and quite ironically, the original reason given for the establishment of the protected area, namely to remove the negative influence of grazing and fire on water production (Pooley 2012), was not influenced by the protected area. Rather an unforeseen or understated threat at the time of establishment was in fact avoided, namely, increased water capture and storage in dams.

The analysis of flow duration curves for the period 1950 – 2000 clearly shows the effect of the 1949, 1972 and 2014 low burn scenarios outside the protected area on streamflow conditions. For these scenarios, sub-catchments outside the protected area had far lower streamflow under mid and low flow conditions in comparison with the natural scenario. This reduction in streamflow increased in magnitude from the 1949 scenario to the 1972 scenario and to the 2014 low burn scenario. Furthermore, declines in streamflow for the 2014 low burn scenario outside the protected area were largest during dry years as opposed to wet years. These results contrast with inside the protected area where reductions in mid to low flow conditions were evident for 1949 and 1972 scenarios but not for the 2014 low burn scenario which showed no significant deviation from the natural scenario of flow conditions. This highlights the importance of the protected area in avoiding streamflow losses and largely points to the influence of dams on reducing streamflow conditions in all sub-catchments outside the protected area.

The negative influence of land use changes, in particular water withdrawal and dam construction, on streamflow over the last four decades has been reported for many case studies in Southern Africa and globally. Fanta et al. (2001) observed declines in streamflow in southern Africa countries starting from about 1975 while Snoussi et al. (2007) identified water abstraction, land use change, and damming as the major reasons for decreasing streamflow in African drainage basins. Nguyen et al. (2017) found annual reductions in streamflow of 13-22% in several Australian catchments and attributed this to water captured by farm dams for irrigation. Most dams in Sub-Saharan Africa were established from the 1960s to 1980s, with a resurgence in the number of dams from the 2000s (Schäfer et al. 2016). This corresponds well with the findings for areas located outside of the protected area in this study. Schäfer et al. (2016) found that 51% of 117 case studies in Southern and Eastern Africa showed decreasing trends in annual streamflow between 1970 and 2010 with the causes attributed chiefly to anthropogenic influences as opposed to being climate related. Schäfer et al. (2016) found distinct regional and temporal differences regarding reported changes and causes. For example, land use change was the main influence on streamflow reported in Eastern Africa, while the construction of dams was the main influence in Southern Africa.

5.4.2 The role of fire and vegetation cover on streamflow

Like most Mediterranean-type ecosystems, fynbos landscapes are prone to fires, which dramatically modify land cover for a number of years (Lindley et al. 1988). In this study, inside the protected area, there were no significant deviations from the overall streamflow conditions under the 2014 low burn scenario. However, this scenario showed declines in streamflow under all streamflow conditions outside the protected area. This highlights the positive role that the protected area has played in reducing dam building activities that could have materialised inside the protected as seen for areas outside the protected area. However, all other land use/cover scenarios and in particular the 2014 high burn scenario increased streamflow under high flow conditions and resulted in subsequent declines under mid-range and low flow conditions both inside and outside the protected area.

These results are in agreement with the majority of research on the hydrological role of vegetation which has extended over the last century (see Brown et al. (2005) and Moreno & Oechel (2012) for reviews). Greater vegetation biomass is considered to have positive effects on streamflow by reducing evaporative loss directly from the soil surface and by slowing surface water and sediment flows. Greater biomass also provides more litter and a habitat for soil fauna, which in turn maintain soil structure and stabilise soils. The water that is retained in the soil is used by plants to transpire which in turn influences vegetation cover. The rest of the water percolates down to the water table and is slowly released to maintain streamflow (Cornet et al. 1993; Aguiar & Sala 1999; Le Maitre et al. 1999, 2014; Brauman et al. 2007). Vegetation degradation exposes more soil directly to evaporation and to the impact of rain at the soil surface. Infiltrability may be further reduced by crusting of the exposed soil, while trampling by livestock may further reduce the time it takes for rainfall to become overland flow as well as the amount of water infiltrated before stormflow commences. This leads to an increased "flashiness" of the catchment with resultant enhanced stormflows while the baseflow store is replenished to a lesser extent than when greater cover of vegetation is present (Scott & Van Wyk 1990; Scott 1993, 1997; Schulze 2000).

The results of this study are specifically in consonance with the existing body of literature on the influence of fire on streamflow in fynbos catchments (Van Wilgen & Kruger 1985; Bosch et al. 1986; van Wilgen et al. 1992; Brown et al. 2005; Moreno & Oechel 2012). The increase in streamflow for the 2014 high burn scenario was ~17-18% greater outside and inside the protected area when compared with the natural scenario. Lindley et al. (1988) found increases in annual streamflow of 15% after prescribed burns in mountain fynbos near Paarl, and Scott (1997) and Scott (1993) found increases of 12-15% after a wild fire in mountain fynbos in Jonkershoek mountain catchments.

The 2014 high burn scenario was composed largely of the high intensity burn area hydrological response unit, which was modelled to reflect post-fire soil hydrophobicity and with less vegetation and surface litter cover in comparison with the natural scenario. The grazing and low intensity burn area hydrological response units were similar to this but were modelled to reflect higher vegetation cover and less soil hydrophobicity, although compaction was accounted for under the grazing hydrological response unit. The reduced capacity for regulating flows is evident from the sensitivity results, which comprised these three hydrological response units and showed that any increase in overall streamflow for the 2014 high burn scenario was largely composed of stormflow. Baseflow and transpiration was reduced while soil evaporation was increased. The sensitivity analyses also showed increases in soil evaporation and stormflow for the grazing and low intensity burn area hydrological response units. Therefore, the grazing hydrological response unit was influential for increasing streamflow under high flow conditions under the 1949 and 1972 land use/cover scenario. However, the low intensity burn area hydrological response unit comprised only a small area under the 2014 low burn scenario and therefore did not play a role on changes in streamflow modelled.

Increases in streamflow for the 2014 high burn scenario in this study only occurred under high flow conditions and resulted in subsequent large declines in streamflow under mid to low flow conditions. Increased high flows and reduced low flows are a typical indication of a mountain catchment with poor capacity for maintaining streamflow throughout dry periods. If the current extent of relatively large fires increases in frequency in the mountains, (i.e. if large areas are being burnt in high intensity burns <10 years apart), this could be detrimental to the capacity of the mountain catchment to regulate streamflow. The consequence of changes to the fire regime could lead to large-scale and rapid reductions in vegetation cover which have both short-term and long-term effects on canopy structure. This could result in substantial reductions in low flow conditions as well as increases in near-ground solar radiation, which in turn drives numerous ecohydrological processes (Royer et al. 2010). However, the effects of disturbance can be complex and can differ depending on the type and spatial coverage of a disturbance to an environment. Although increases in stormflow have been reported for reductions in vegetation cover and near-ground solar radiation generally increases with reductions in vegetation canopy cover, specific characteristics of such trends that may be potentially important in an ecohydrological context have not been directly assessed (Royer et al. 2010; Wang-Erlandsson et al. 2014). Due to the static nature of the modelling undertaken in this study, further works is required to fully understand the influence of changing fire patterns on streamflow.

5.4.3 Potential modelling priorities

Despite extensive research, there is still debate over the hydrologic response to perturbations in catchments in semi-arid environments, and even larger uncertainty exists when scaling up to regional scales and over longer periods of time (Zhang et al. 2001; Meixner & Wohlgemuth 2003; Brown et al. 2005; Guardiola-Claramonte et al. 2011; Moreno & Oechel 2012; Van Dijk et al. 2012; Dye 2013). Changes in annual streamflow from paired catchment studies in fynbos catchments have been shown only for the first 1-4 years post fire (Le Maitre et al. 2014). In this study, the system was modelled in a static state, i.e. 1-4 years after a high intensity burn, and therefore only these initial increases in streamflow were accounted for in the modelling process for the 2014 high burn scenario. Although studies have shown that fires can cause immediate large increases in streamflow in certain contexts the longer term impacts, i.e. 50 to 100 years, of a change in fire regime on the hydrological system have not been fully investigated (Stoof et al. 2012; Kinoshita & Hogue 2015; Bart 2016) especially in Mediterranean shrub environments such as fynbos. Furthermore, studies have not fully considered ecohydrological factors associated with the partitioning of water flows and how this influences changes in streamflow (Guardiola-Claramonte et al. 2011).

Modelling efforts require a sound conceptual understanding of how vegetation cover influences the individual components that make up the different portions of the water budget in a catchment (Bulcock et al. 2014). Due to the static nature of the modelling undertaken in this study, vegetation recovery during post-fire conditions was not accounted for in the modelling process. Furthermore, the natural scenario did not include the natural fire regime for the fynbos biome. Therefore, this is not a correct representation of the long-term, natural flow conditions. While there are limited examples of paired catchment studies which examine the impact of permanent vegetation changes on water yield, different responses have been recorded in different regions (Brown et al. 2005). Given the projected increases in temperature and the possible interaction with fire regimes in Mediterranean type environments, the influence of changes in vegetation cover on the different parts of the water budget and how this manifest throughout the hydrological regime is an area for future research. This is especially important for modelling efforts in ungauged catchments (Savenije 2002; Bulcock et al. 2014). While there have been a number of empirical studies on streamflow, far fewer have been able to partition evapotranspiration losses into soil evaporation and transpiration processes (Zhang et al. 2001; Guardiola-Claramonte et al. 2011).

Ideally, in future studies, fire and other disturbances should be dynamically implemented in the modelling process to account for time dependent changes. Given that fynbos is a fire driven biome incorporating this would be critical to understand long-term impacts and consequences of land use/cover change and protection. Probabilistic patterns of changes in climate and land use/cover also require better integration for understanding the influence on water flows. Ideally, for future work, ACRU or another suitable hydrological model should be run in a dynamic mode in which the spatial coverage of hydrological response units as well as the respective land use/cover-specific variables change dynamically within the model over the time series. In addition, modelling of water storage needs to be improved based on additional fine scale information and routines incorporated in the model. It is likely that this would result in more realistic modelling of the impact of land use and cover, and especially fire, on hydrology.

5.5 Conclusion

This study contributes to understanding the impact of mountain protected areas on land use/cover, fire and streamflow by considering several plausible counterfactual arguments. The protected area was found to have considerable influence in terms of protecting water supply, particularly during dry years. The avoidance of dam building in the protected area was an important factor in maintaining natural streamflow conditions. However, changing fire patterns both outside and inside the protected area have influenced streamflow responses in ways that could not have been foreseen when the protected area was established. In addition, despite the positive influence of the protected area on water supply downstream there have been local ecosystem service trade-offs associated with direct and indirect lost opportunities for socio-economic development within the protected area. This highlights the importance of multiple land management system types within mountain systems including protected environments as well as areas for sustainable land practices to support local socio-economic development such as multifunctional landscapes.

The study highlights the need for enhanced modelling efforts in the Cape Floristic Region to evaluate the hydrological response of changing land use/cover, especially associated with fire and climate but also infrastructure such as dams. This requires empirical data on water fluxes from vegetation cover types in the biome coupled with modelling exercises, which incorporate dynamical and probabilistic operations related to changes in climate as well as land use, fire and vegetation cover and improved configurations of water use in catchment areas. Many watershed studies focus on direct responses in water yield following vegetation change (Meixner & Wohlgemuth 2003). There is a need for more empirical studies that can be linked to modelling studies to better understand longer term and permanent changes in water flows in response to climate change and variability and mediated through land use and vegetation change.

Section 6. General discussion and synthesis

In this section, I firstly summarise the main arguments and findings of the thesis. I then discuss the broader significance of the research and identify research needs. I conclude the thesis by reflecting on the use of a pluralistic, socio-ecological approach from an individual researcher perspective.

6.1 Main arguments of the thesis

In this thesis, I provide clear evidence of the dynamic nature of land use/cover and associated ecosystem service trade-offs in relation to protected area establishment. I show the prominence of interactions between global and local drivers that ultimately determine the influence of a protected area on land use/cover and water-related ecosystem services. In doing so, I emphasise the importance of using a counterfactual approach to understand the effectiveness of conservation actions, as has also been noted by recent studies in the protected area impact evaluation literature. However, I highlight the need to use a pluralistic, socio-ecological approach for developing counterfactual conditions to obtain a full understanding of the impact of protected areas at a place-based level and especially within mountain ecosystems.

On first instance, I confirm that strict protection in mountain areas can prevent an increase or intensification of land use/cover types associated with infrastructure, e.g. buildings, dams, roads and cultivation, and thereby can have a positive influence on the release of water-related ecosystem services. However, understanding who benefits from this protection in terms of human well-being is something that requires more research. For example, I show that if the protected area had not been established it is likely that water flows may have been stored for local use, but because the protected area was established the water is released for downstream use. Most of the water leaving the catchment is used for domestic and economic use in urban centres in the Western Cape via the Western Cape Water Supply System. In contrast, outside the protected area, on mountainous land of similar terrain, the socio-economic conditions have improved with more opportunities such as employment associated with agriculture and ecotourism. These local socio-economic benefits have been prevented inside the boundaries of the protected area. Therefore, it is important to appreciate that multiple management system types are needed depending on the socio-ecological setting and that these should be informed by considering local, regional and global trade-offs to promote sustainability.

6.2 A summary of the main findings for the mountain case study (see Table 6.1)

6.2.1 Land use/cover change

This study showed that for outside the protected area there was a shift from subsistence use of mountain land for grazing and small-scale farming to personal nature-based recreation and ecotourism with cultivation intensified in suitable areas of mountain land. The abandonment of mountain land for grazing resulted in an increase in the area covered by natural or semi-natural vegetation. However, intensification of cultivation and an increase in ecotourism was accompanied by an increase in infrastructure. This included exponential increases in buildings and dams as well as linear increases in roads. Had there not been protection, it is likely that all these changes would have occurred inside the currently protected area.

Global socio-economic pathways were identified as a main driver of the abandonment of mountain land for grazing outside the protected area since the mid-20th century. The area used for grazing declined from 34% to 8% and the number of landowners that used their mountain property for grazing declined from 41% to 20%. Global socio-economic pathways were also important for driving changes in wildflower harvesting and the substantial increase in personal nature-based recreation and ecotourism outside the protected area. There was a 61% increase in the number of landowners using their mountain land for personal nature-based recreation and a 23% increase in ecotourism. Increased productivity occurred in flat or valley areas due to improvements in technology, together with the inability to modernize mountain land, due to land steepness or poor access. Major influences included the exchange of and exposure to new commodities, markets, ideas, technologies and conservation ideologies that resulted in complex upland-lowland transitions and related socio-ecological feedbacks.

6.2.2 Fire

Global drivers, which resulted in the abandonment of land for grazing, interacted with national level fire policy to bring about a reduction in the use of fire in the Groot Winterhoek Mountains. Based on the analyses of orthoimages over the last ten years it was evident that the current landscape exists either in a state in which few or no fires occur (low intensity fire regime) or in a state in which a few large, intense fires occur (high intensity fire regime). This is very different to the patchy fire regimes of the past which promoted a mosaic of post-fire vegetation ages in the landscape. Importantly, changes in fire patterns occurred regardless of protected area boundaries. Therefore, in the absence of the protected area these changes would have also occurred inside the area currently protected.

6.2.3 Vegetation

The combined decline in grazing and frequent patch burning in the Groot Winterhoek Mountains resulted in a local-level vegetation response, notably an increase in basal and canopy vegetation cover. This is in line with existing farmland abandonment research which shows that a higher risk of starting and propagating fires because of increased plant biomass is recognised as one of the more obvious effects of farmland abandonment, especially in Mediterranean zones (Lasanta et al. 2015). The findings suggest that the elimination of grazing and frequent small and cool fires of the past was the driving factor that caused the increase in vegetation cover over the last four decades (Van Wilgen 1982; Van Wilgen et al.

1994; Vlok & Yeaton 2000). The vegetation age of sites sampled in this study prior to the most recent fire was between 14-24 years, which is ideal for fynbos sustainability (van Wilgen et al. 2016a). Therefore, the current wildfires and anthropogenic accidental fires in the landscape have been beneficial for the vegetation in the protected area in comparison to baseline measures.

However, changing fire patterns were experienced regardless of protected area boundaries. Therefore, in terms of the counterfactual argument regarding the impact of the mountain protected area, the positive change in vegetation cover cannot be attributed to the establishment of the protected area alone. Rather, increased opportunities and better living standards associated with lowland areas, increased globalisation, increased access to the market economy and improved economic growth and greater infrastructural development caused these shifts that ultimately influenced vegetation cover inside the protected area. Fragmentation of existing natural vegetation, however, has been avoided inside the protected area.

6.2.4 Streamflow

The increase in vegetation cover experienced in the mountains, due to the reductions in grazing and burning that were prolific prior to the 1970s, was modelled to show an improvement in the catchment's ability to maintain streamflow during all flow conditions. This however, only occurred inside the protected area and only under the low intensity fire regime. These positive changes in streamflow were not present outside the protected area where streamflow was reduced under all flow conditions in comparison to a natural scenario of streamflow under a low intensity fire regime.

Negative impacts to overall streamflow conditions were likely due to the exponential increase in water stored and used for irrigation and other uses outside the protected area from the 1970s. This was avoided inside the mountain protected area primarily because cultivation was removed, and no dam building occurred. Therefore, the protected area proved significant in avoiding reductions in streamflow particularly so during mid to low flow conditions and dry years. However, this positive influence was confounded by the influence of changing fire patterns. When the current land use and cover scenario was modelled under a high intensity fire regime it resulted in negative effects on the ability of the mountain catchment to maintain streamflow both outside and inside the protected area.

Therefore, under a low burning regime scenario, mountain protection had considerable influence on maintaining streamflow, particularly during mid and low flow conditions and during dry years, i.e. when water is most needed for human consumption.

6.2.5 The value placed on mountain protection by landowners

Landowners outside the protected area mostly placed intrinsic and non-use values on the mountain protected area as opposed to direct use such as recreational or spiritual values. The only other use value that scored as high was the indirect use value of water supply. There was also a relationship between access to nature-based recreational activities outside the protected area and finding merit in mountain-protection. Personal nature-based recreation was also associated with intrinsic and non-use values of mountain protection. Therefore, although greater fragmentation of vegetation cover and increased water storage and use occurred outside the protected area due to the increase in roads and other infrastructure related to intensification of cultivation and ecotourism. This allowed for greater accessibility of mountain areas for nature-based recreation in comparison to inside the protected area and an improvement in socio-economic opportunities which were not equally realised inside the protected area. This highlights the importance of multi-use zones in a mountain landscape for ecosystem services but also the need for greater information on how these ecosystem services are apportioned between social groups both temporally and spatially.

Table 6.1 Temporal and spatial trends in ownership, land use/cover, fire, vegetation and streamflow before and after protection in 1978 for areas outside and inside the currently protected mountain area. Findings summarise results from landowner surveys and in-depth interviews, repeat vegetation surveys and ground terrestrial photographs, historical and current orthoimage analyses and hydrological modelling of land use/cover scenarios.

SPACE	TIME Mountain protection	BEFORE PROTECTION (1940-1970s)	PROTECTED AREA ESTABLISHED IN 1978	AFTER PROTECTION (>2010s)
OUTSIDE PA	Ownership	• Privately owned land.		• Privately owned land.
	Land use/cover	• Burning for grazing and wildflower harvesting. • Extensive cultivation in suitable areas of mountain land. • Roads, buildings, dams and scattered alien trees near settlements • An increase in dense alien plantations, cultivation and dams during the 1970s. • Reduced natural vegetation cover due to grazing accompanied by burning.		• Substantial declines in the area used for grazing of ~26% and wildflower harvesting of ~13%. • Cultivation intensified in areas of suitable mountain land with the number of landowners using their mountain land for cultivation increasing by 20% but with only marginal increases in the area under cultivation. • Exponential increases in buildings and dams since the 1970s, and linear increases in roads along with increased socio-economic opportunities. • Diversified ecosystem service use, including an increase in landowners using their mountain land for personal nature-based recreation of ~61% and ecotourism of ~23% along with increased socio-economic opportunities. • Dense alien plantations have been removed with scattered alien trees reduced in comparison to the 1970s. • Increased natural vegetation cover due to declines in grazing and burning.
	Fire	• Small, frequent and cool (low intensity) fires due to use of burning for grazing and the presence of low fuel loads. • Patchwork of fire scars with an average fire size of 30 ha during the 1940s. • During the 1940s, >100 fire scars over a five-year period with one third of the area being burnt (~5200 ha/32% of the area burnt). • Fires become larger during the 1970s with an average fire size of 144ha. • The frequency of fire scars also declines into the 1970s with ~30 fire scars over a five-year period but with the total area burnt remaining similar (~4100 ha/~25% of the area burnt).		• Landscape switches between a high and low intensity-fire regime due to the limited use of burning and accumulation of high fuel loads. Certain landowners have recently attempted to implement controlled burning. **High intensity fire regime** • Large, less frequent then before protection, but high intensity fires. • Large areas of vegetation cover burnt with an average fire size of ~3080 ha. • ~4 fire scars over a five-year period burning almost all the vegetation (~12 300 ha/~78% of the area burnt). **Low intensity fire regime** • High intensity fire periods are followed by years of smaller, less frequent then before protection, low intensity fires • Fire sizes are on an average larger than the 1940s but similar to the 1970s (~90 ha) but burning very limited portions of the catchment due to declines in frequency. • ~2 fire scars over a five-year period burning very limited portion of the catchment (~270 ha/~2% of the area burnt).
	Vegetation	• Patch mosaic of vegetation ages ranging between 1 and 5 years. • Low vegetation cover and open canopies. • Fragmentation of vegetation due to land use and cover present including grazing paths, roads, cultivation and related infrastructure. • Pockets of very old vegetation exist in certain areas due to the lack of suitability for grazing and biophysical conditions excluding fire.		• Increased fragmentation of vegetation due to increases in buildings and roads and related infrastructure. **In areas without fragmentation** • Homogenous vegetation age across large portions of the mountain catchment varying from 1- 15 years. • The landscape switches from drastically reduced vegetation cover for approximately 1-5 years after high intensity fires to good vegetation recovery and high vegetation cover after 5 years. • Increased vegetation cover compared to before protection with closed shrub canopies after 5 years of vegetation recovery after high intensity fires. • Pockets of very old vegetation due to biophysical conditions excluding fire.
	Streamflow	• Reduced streamflow of ~2 to 7% during mid to low flow periods and increases of ~11% during high flow conditions in comparison to natural streamflow conditions due to the removal of vegetation and changes to soil structure from grazing and burning during the 1940s.		**Low intensity fire regime** i.e. five years following substantial vegetation recovery after large and high intensity fires • Reductions in streamflow across all flow conditions (high, mid and low) and especially during dry years, despite an increase in impervious areas and the decline in dense alien trees. • Reductions range from 1% under high flow conditions to 25% under low flow conditions with reductions increasing in magnitude during dry years as opposed to wet years.

		• Increases in streamflow consist mainly of flashy overland flow with reduced baseflow. There is also increased soil evaporation while transpiration is reduced. • These streamflow patterns continue into the 1970s, but declines increase under mid and low flow to ~9 and 23% respectively due to the increase in dense alien plantations and cultivation as well as minor increases in dams. • Increased streamflow under high flow conditions persists due to the increase in impervious areas and continued grazing and burning. • There is no relationship between reductions or increases in streamflow and total annual rainfall.	• Any increase in flow expected from an increase in impervious areas and the decline in dense alien trees has been dampened by the exponential increase in dams and the intensification of irrigation practices. **High intensity fire regime** i.e. within the first five years after high intensity fires • A loss in the capacity of the catchment to maintain streamflow with extreme increases (~80%) under high flow conditions and reductions (~30-60%) under mid to low flow conditions. • Excessive increase in stormflow and resulting loss in baseflow, an increase in water lost through soil evaporation as opposed to transpiration. • Greater losses of ~14% under low flow conditions in comparison to inside the protected area despite less area being burnt in high intensity burns due to additional losses caused by land use activities.
INSIDE PA	**Ownership**	• Privately owned land.	• State owned land.
	Land use/cover	• Burning for grazing and wildflower harvesting. • Extensive cultivation in suitable areas of mountain land. • Roads, buildings, dams and scattered alien trees near settlements. • Reduced area covered by natural vegetation cover due to grazing accompanied by burning.	• Highly regulated and permit-controlled nature-based recreation. • Grazing, wild plant harvesting, and cultivation no longer occur except for some poaching of certain wild plants. • Some access roads and buildings but with reduced area in comparison to before protection. • Limited scattered alien trees reduced in comparison to before protection. • Increased area covered by natural vegetation cover due to declines in grazing accompanied by burning. • Reduction in socio-economic opportunities associated with agriculture and ecotourism
	Fire	• Small, frequent and cool (low intensity) fires due to the use of burning for grazing and the presence of low fuel loads. • Patch work of fire scars with an average fire size of 30-35 ha. • >100 fire scars over a five-year period with a third of the area being burnt (~5000-5300 ha/34% of the area burnt). • Large fires are the exception in the landscape.	• Landscape switches between a high and low intensity-fire regime due to the limited use of burning and accumulation of high fuel loads. **High intensity fire regime** • Large, less frequent then before protection, but high intensity fires. • Large areas of vegetation cover are burnt with an average fire size of ~4700 ha. • ~3 fire scars over a five-year period burning almost all the vegetation (~14 300 ha/92% of the protected area burnt). **Low intensity fire regime** • High intensity fire periods are followed by years of smaller, less frequent then before protection, low intensity fires. • Fire sizes are still larger than before protection with an average fire size of ~123 ha. • ~2 fire scars over a five-year period burning very limited portion of the catchment (~250ha/~2% of the protected area burnt).
	Vegetation	• Patch mosaic of vegetation age ranging between 1 and 5 years. • Low vegetation cover and open canopies. • Fragmentation of vegetation due to land use including livestock paths, roads, cultivation and related infrastructure. • Pockets of very old vegetation exist in certain areas due to the lack of suitability for grazing and biophysical conditions excluding fire.	• Homogenous vegetation age across large portions of the mountain catchment varying from 1- 15 years. • The landscape switches from drastically reduced vegetation cover for approximately 1-5 years after high intensity fires to good vegetation recovery and high vegetation cover after 5 years. • Increased vegetation cover compared to before protection with closed shrub canopies, after 5 years of vegetation recovery after high intensity fires. • Reduced fragmentation of vegetation in comparison to before protection. • Pockets of very old vegetation in the catchment due to biophysical conditions excluding fire.
	Streamflow	• Reduced streamflow of ~2-3% and ~5% during mid to low flow periods respectively and increases of ~8-9% during high flow conditions due to the removal of vegetation and changes to soil structure from grazing accompanied by burning. • Increases consist mainly of flashy overland flow with reduced baseflow. • Increased soil evaporation while transpiration is reduced.	**Low intensity fire regime** i.e. five years following substantial vegetation recovery after large and high intensity fires • Streamflow recovered and well maintained under all flow conditions, as well as across dry and wet years. • No difference to what would be expected under natural streamflow conditions. **High intensity fire regime** i.e. within the first five years after high intensity fires • A loss in the capacity of the catchment to maintain streamflow with extreme increases of streamflow of ~80% under high flow conditions and reductions of ~30 to 44% under mid to low flow conditions. • Excessive increase in stormflows and soil evaporation and resulting extreme losses in baseflow with reduced transpiration.

6.3 Broader significance of findings

In this thesis, I contribute conceptually and theoretically to our understanding of the dynamics of ecosystem services in relation to mountain protection. Methodologically, I demonstrate an inclusive research approach for improving evidence-based conservation at a place-based and landscape level. I highlight the importance of linking protected area impact with societal perceptions and preferences within the overall goal of creating an evidence base for conservation to support protected area persistence given an uncertain future.

6.3.1 Evidence-based conservation: protected area impact versus persistence

Regardless of efforts made to integrate protected areas into a wider landscape, many authors suggest that protected areas are still managed as islands within a matrix of degraded territory with no clear conceptual framework to integrate them into the surrounding social landscape (e.g. agricultural, business and cultural landscapes). This isolation is perceived by many as a risk to the ability of protected areas to persist into the future and has the potential to influence the position of protected areas negatively when competing with other land use demands (Palomo et al. 2014; Cumming 2016).

Determining protected area impact should only be considered as one part of developing an evidence base for conservation. The second part is matching impact with societal preferences and creating awareness on protected area impacts that may not be valued by society or decision makers (Bennett 2016). There will always be trade-offs required for stimulating support for traditional conservation areas, and in this regard, there is a need to increase the appreciation for the potential role that conservation authorities could play in developing, protecting and maintaining novel and hybrid ecosystems in accessible areas for stimulating revenue and societal support for more traditional biodiversity focused protected areas. It would be in these areas that awareness could be raised of the benefits, such as water supply, provided by protected areas in difficult to reach places.

A number of studies have considered the importance of direct use values, such as recreation or spiritual values inside protected areas, as mechanisms for maintaining or garnering support for protected areas (Chan et al. 2006; Daniel et al. 2012; Maciejewski et al. 2014; Ament et al. 2016). However, few have considered the role of intrinsic or non-use values in connecting people to nature and for providing support for protected areas (Vucetich et al. 2015). It is well accepted that nature-based tourism in protected areas offers an important connection between protected areas and society (Maciejewski et al. 2014). However, not all individuals are in the position to access protected areas or private natural areas for recreational based activities.

Intrinsic and non-use values for protected areas require no direct use of a service from a protected area but could require exposure, experience or alternatively some kind of link to the natural environment. For example, nature-based recreation may be similarly achieved in semi-natural habitats, parks with uniform grass, or mixed, multi-use, forest landscapes. Although these novel landscapes may be more appealing in terms of their aesthetics to certain societal groups as opposed to strictly natural areas, these landscapes may not fulfil the conservationist's vision of protecting biodiversity and ecosystem services (Schröter et al. 2017). However, the desire for nature-based recreation within novel ecosystems could provide an opportunity for conservationists to engage with broader dimensions of society for creating awareness about the importance of "less aesthetically" appealing landscapes. Without understanding the values and views placed on different types of conservation areas by society it will be difficult to harness the support required for ensuring the persistence of conservation areas especially given an uncertain future (Bennett 2016; Cumming 2016).

6.3.2 Evidence-based conservation: plurality in methodologies

Studies that assess future ecosystem services are commonly based on a set of ecosystem services that are currently considered important by the stakeholders or the researchers. This study, however, shows that demand for ecosystem services changes over time and therefore current desired ecosystem services may not persist in future. Furthermore, it is challenging to predict what ecosystem services will be recognized as important by society in future (Bürgi et al. 2015; Tomscha & Gergel 2015; Tomscha et al. 2016).

In this study, remote sensing, interpretative social science approaches, scenario planning and hydrological modelling all provided important quantitative and qualitative linking methods for incorporating multiple methods, knowledge and information sources into the investigation of protected area influence. Increased historical temporal coverage was achieved through historical aerial photographs and social surveys and interviews amongst other methods that aimed to provide fine scale place-based information on ecosystem service changes over time. Empirical evidence generated from the analyses of repeat vegetation surveys, terrestrial oblique photographs, aerial orthoimages and quantitative landowner survey questions provided quantifiable measures of change as well as associations. Qualitative in-depth landowner knowledge on the other hand allowed for capturing the complexity inherent in socio-ecological system dynamics such as cross-scale dynamics and threshold effects and thereby allowed for causal inference for quantitative measures.

Social science methodologies were critical for the analysis of causal pathways and the interplay between complex system components. The modelling process allowed for the incorporation of all data including empirical and social science qualitative data and interpretative analyses to understand change in streamflow. Considering counterfactual thinking within the overall pluralistic, socio-ecological approach proved critical for determining the role and challenges of conservation in the landscape. Without explicit attention to counterfactual analyses, change in threats and associated ecosystem service impacts could have been misaligned with protected area impact (Pressey et al. 2015). However, the integration of multiple approaches combined with interdisciplinarity, as well as disciplinary and temporal depth were critical to achieve an understanding of socio-ecological change as well as to understand the impact of protection within a socio-ecological context.

To predict the impact of conservation under future conditions in an ethical and effective manner, there is a need for further work that investigates how datasets and findings from retrospective studies can be used to inform a future scenario building and modelling approach. Creative techniques should be used to forecast what types of ecosystem services will be important under future conditions and how these will be apportioned between different social groups both temporally and spatially. Varied approaches can be used and blended to understand past threats as well as to anticipate future threats. Statistical models could be used together with the knowledge of stakeholders and more in-depth ethnographic accounts to predict future investments based a range of global and local circumstantial scenarios. These results would provide an indication of the future distribution of conservation threats, which could be linked to protected area impact (Haruna et al. 2014).

6.4 Research needs

Identifying those who benefit from ecosystem services and understanding how benefits are distributed among individuals and stakeholder groups is an acknowledged prerequisite for effective ecosystem service assessments (Bennett et al. 2015). What is missing from this study are the perceptions that local communities that live in nearby towns or domestic water users that benefit from Cape Town's Water Supply System have towards mountain-protection. For example, in this thesis there was limited investigation of the ecosystem service trade-offs that have occurred between upstream and downstream areas and how these have influenced different types of social groups both spatially and temporally.

Landowners interviewed in this study form part of households that largely fall within the commercial farming districts of the Western Cape. Despite the past reliance on mountain land for subsistence purposes, they currently use the natural environment directly for non-subsistence provisioning services and for cultural ecosystem services. They also source most of their natural resources through indirect interactions with the environment. Further research is required to understand the diversity of stakeholder views at different spatial-temporal scales and how this links to the local versus regional and global trade-offs that result from protecting natural environments. This gap is not unique to this study, as it also remains largely unaddressed in the international research agenda (Bennett et al. 2015). Additional work is required in valuing ecosystem services and the role of conservation, in individual, social, community, and group contexts (Costanza et al. 2017). A direction for future research could be a more ethnographic exploration of the role of protected areas and land use and cover on ecosystem services changes in the area and how these have been influenced by different social, political and economic factors. There should also be a focus on determining appropriate pluralistic approaches for aggregating placed-based research at regional and global levels for consolidation and higher-level analyses.

Understanding increases in streamflow under high flow conditions due to fire in mountain catchments is of obvious importance to assist decision makers in terms of extreme risk avoidance, e.g. flooding and land slips. However, it is the longer-term impacts on baseflow and low flow conditions that may be of greater concern in terms of water security (Meixner & Wohlgemuth 2003). The latter is especially concerning since surface water is a key resource, and one that is already over-exploited in South Africa. Water scarcity is most pronounced and shows the strongest effects in times of droughts, which recur frequently in South Africa (Schäfer et al. 2016). Furthermore, it is likely that water demand in the winter-rainfall part of the Western Cape will only increase because the growing season for many crops coincides with the dry summer months (New 1999, 2002).

There is a need for research on the influence of changing land use/cover patterns on hydrology and water security especially in relation to changes in climate conditions associated with increased drought events. Future empirical work is required for understanding the role of vegetation cover on water fluxes in mountain ecosystems and extrapolating this to catchment scales. A focus should be on the influence of changing fire regimes in upland catchments, including incorporating dynamical and probabilistic inputs into modelling routines for understanding the influence of land use/cover as well as climate on hydrological systems. Additional modelling studies and empirical ecohydrological studies would assist to develop a full understanding of the influence of the dynamic nature of changing fire regimes on the provision of water from water source areas in mountains.

Presentation of uncertainty in modelling results also requires more transparency i.e. the influence of the model to different land use and cover parameterisations. In addition, there is a need to increase the accuracy of model configuration in terms of dams and water storage and use. This study used an approach whereby all dams are lumped together at the end of the sub-catchment. This is a similar approach used in other South African studies that have used ACRU to model the influence of land use/cover on streamflow (Warburton et al. 2012). Although seen as a standard approach, this is not entirely realistic, and could be influencing the results of the study. A more thorough understanding of water capture and use in this and other catchments and an improved representation within model configurations is required to obtain an accurate reflection of the influence of water storage in mountain catchments on downstream water supply.

6.5 Reflections on the usefulness and challenges of a pluralistic, socio-ecological approach

The integration of multiple types of methods into protected area impact evaluations leads to a greater understanding of the complex social, political, and economic contexts within which conservation occurs. All of these information types can be used to guide or improve conservation policies, management actions, and ecological outcomes (Bennett 2016). However, it is acknowledged that using integrated research for generating evidence for informing conservation interventions is not without difficulties (Rissman & Gillon 2017). Most of the literature on integrated research focuses on using teams to achieve trans- and interdisciplinarity (Pooley et al. 2014). Team members work on specific or disciplinary components of a project and then a different individual or group of individuals works on the integration aspect. Problems largely centre on communication difficulties between team members as well as conflict arising from differing mental models among various members in these teams. This is largely because mental models are different between different types of disciplinary focused academics and secondly mental models differ among conservation scientists and managers and general society representatives.

These above specific challenges were not experienced in this current thesis. This is because all disciplinary and integration components were undertaken by an individual researcher. The pluralistic, socio-ecological approach enabled me to prioritise the input from stakeholders (managers and landowners) by incorporating their views in the beginning of the research process. It also enabled me to use these inputs to identify relevant disciplines and thus bridge disciplinary boundaries. All of this was achieved without the usual challenges that exist in trans- and interdisciplinary team settings. This is because I was largely working within the realms of my own mental model and therefore conflict was limited in comparison to working within team settings. However, multiple challenges were still present throughout the research process. These were mostly associated with differing cultures and epistemologies within the university environment and funding and operational costs.

Attempting to communicate the relevance of methods and theory from across disciplines to discipline-specific individuals was challenging. In this way, disciplinary and institutional boundaries still placed strain on the inclusion of a broad range of data, knowledge and analysis types. In most cases, there was more appeal for results separated out into their respective disciplinary domains than broad overarching integrated results. The preference was also for quantitative over qualitative data. Landowners and managers also differed in their views on what questions and results were interesting and applicable in comparison with what was viewed as necessary for doctoral research. This creates strain on the transdisciplinary process whereby certain aspects are not included due to academic structures, which interplay with funding and operational cost barriers. For example, while disciplinary input is critical to achieve disciplinary rigor it is also detailed and, in most cases, fine scaled at a disciplinary level. Given funding and time constraints it is difficult to address all disciplinary requirements while also focusing on overarching applied research questions within a trans- and interdisciplinary context. In this regard, the lack of mutual understanding of the theory and methods across disciplines limits the feedback that can be received at a broader integrated level. Certain academics are generalists while others are specialists within specific disciplinary boundaries and harnessing the strengths of both these types of academics is important. Although an oxymoron, there is definitely more scope within universities for developing academics that are "trans- and interdisciplinary specialists" or "generalised specialists".

Regarding a complete pluralistic approach there is only a certain level of depth and justice that can be achieved from an individual researcher or even within a team setting. Furthermore, ways to increase "sample sizes" and spatial scales, i.e. regional and global levels of analyses, while upholding pluralism for evidence-based conservation requires further consideration. While place-based and landscape levels of analyses are critical, there is also a need for both higher levels of analyses as well as more ethnographic approaches and this was not achieved in the pluralistic, socio-ecological approach used in this study. Furthermore, the level of change that will result from the research presented in this thesis is unknown. This will depend on continual and close collaboration with social science and governance scholars and engagement with relevant stakeholders and managers. Given the lessons learned and insights from the thesis, a few key principles are proposed that can support a pluralistic, socio-ecological approach at multiple levels of analyses. These include:

- Consider a range of disciplinary methodologies, methods and analytical approaches i.e. embrace interdisciplinarity (Jantsch 1970)
- Incorporate the need for operational boundaries but include multiscale perspectives (Cilliers 2005; Flood 2010)
- Accept the limits and boundaries of knowledge in that it is not possible to give a complete, analytical and formal description of a complex problem (Cilliers 2005)
- Acknowledge the role of humans and in particular values, beliefs and experiences in describing, designing and interpreting change (Cilliers 2005), including society, researchers and decision-makers.
- Combine multiple forms of logic and knowledge for uniting aspects of different traditions between disciplines or groups of society and informing one or more theoretical models (Downward & Mearman 2006; Meyer & Lunnay 2012).
- Recognise that multiple theories exist to describe the causal mechanisms of socio-ecological change and ultimately there are many theories on how society can transition to sustainability and with which a researcher can contextualise their work (Peter & Swilling 2014)
- Focus on implementation and solutions enabled through detail and specification but allow for aggregation of specifics for the purpose of generalisation (Flood 2010)
- Embrace the importance of perceptions (Bennett 2016)

6.6 Concluding remarks

The findings of the thesis highlight the relevance of mountain protected areas for achieving conservation goals. I show how mountain protection can prevent negative land use/cover change related to loss of downstream water-related ecosystem services. I also show the importance of a range of values associated with mountain protection, including intrinsic, non-use values (existence, bequest and option) and indirect use values (water supply). However, the thesis highlights that there are ecosystem service trade-offs associated with protection and that these trade-offs manifest in different ways for social groups both in space and time.

For example, despite the positive influence of protection on water flows for downstream use, the local opportunity costs for the protected portion of the case study have included limited socio-economic development associated with agriculture and ecotourism and thus limited benefits such as employment and increased nature-based recreation experiences.

Many confounding and moderating factors influence whether a protected area has an impact on biodiversity and ecosystem services. These factors are highly context specific but can be influenced by drivers at multiple scales. A rigid approach to understanding protected area influence on socio-ecological outcomes is not in line with building long-term support for conservation. We should strive to obtain an understanding of context, including the global, regional and local ecosystem service trade-offs that result from different types of land management and use systems, and associated societal perceptions and preferences. These should inform a multiple land use strategy approach to conservation. To understand fully the influence of protection on long-term environmental trends within protected areas we clearly need to use counterfactual thinking. However, this needs to be contextualised with integrated research and knowledge and an understanding of the values and views of society and decision makers on protected areas as a tool for conserving ecosystem services.

Appendix 3.1 Landowner survey and guideline for interviews

This questionnaire was used to survey as well as to guide in-depth interviews with landowners to capture landowner and land use characteristics as well as views and values on mountain protection and fire policy. Asterisks indicate repeated questions that were originally asked to landowners in 1978. Responses were captured electronically and therefore space for in-depth responses are not evident in this form.

Exploring land use and fire in the Groot Winterhoek Mountains

Consent statement

I understand the purpose of this study and voluntary agree to participate in this questionnaire and interview process

☐ agree ☐ disagree

Section 1: General

1) Farm/Property name/s

2) Farm/Property number/s

***3) Total area of farm/property within the mountains**

Approximate hectares or square kilometres. If only a portion of your property falls within the mountains please only include the area of this portion. You can estimate this or leave it blank if you are not completely sure.

4) How did you acquire your mountain land?

☐ purchased
☐ inherited
☐ other:

5) How long have you owned your mountain property?

6) Do you live on your mountain property?

☐ no
☐ yes, permanently
☐ yes, for part of the year
☐ yes, for part of each month of the year
☐ no, but a manager stays on the property
☐ other:

7) Do you conduct economic activity on the mountain?

☐ no
☐ yes, my main economic activity
☐ yes, my only economic activity
☐ yes, part of my economic activity
☐ other:

8) Do you have a temperature or rain gauge on your mountain property?

If yes, please indicate how many years of data has been collected and indicate the data quality.

Section 2: Livestock

Section 2.1 Livestock grazing

***1) Is your land used for grazing livestock and what type?**

☐ no
☐ yes, cattle only
☐ yes, sheep only
☐ yes, goats only
☐ yes, cattle and sheep
☐ yes, cattle and goats
☐ yes, sheep and goats
☐ yes, cattle, sheep and goats
☐ other:

*If you answered **NO** to the question above, **MOVE directly TO SECTION 2.2** and skip the questions below*

***2) What is the size of the area used for grazing?**

Approximate hectares or square kilometres (a percent/proportion of the total mountain property area can also be given if easier)

***3) What is the time of pasturing/grazing?**

Please indicate during what period of the year the land is being utilised for grazing

☐ all year round
☐ autumn/winter: 1-4 months
☐ autumn/winter: greater than 4 months
☐ spring/summer: 1-4 months
☐ spring/summer: greater than 4 months
☐ other:

***4) What is the total number of livestock on the property?**
Total number of livestock (e.g. cattle, sheep, or goats)

Section 2.2 Other livestock production systems

1) Are there horses, mules or donkeys on your farm?
MORE THAN ONE OPTION CAN BE SELECTED.
☐ no
☐ yes, horses in a paddock (fenced area)
☐ yes, mules, donkeys or horses as working animals on the property
☐ other:

2) What other types of livestock production occur on your property (if any)?
Are there any other forms of domestic livestock e.g. pigs*? **MORE THAN ONE OPTION CAN BE SELECTED.***
☐ no other types occur on the property
☐ pens with supplementary feed
☐ camps with supplementary feed
☐ other:

If you answered, "NO" and "NO other types occur on the property" above, MOVE directly TO SECTION 3 and skip the questions below.

3) What is the size of the area used for the livestock in pens/camps and/or horses/donkeys/mules?
Approximate hectares or square kilometres, (a percent/proportion of the total mountain property area can also be given if easier).

---

4) What type of livestock is used in the pens or camps indicated in question 2 above?
Include all information that you feel is relevant for the study to describe the livestock production system used.

Section 3: Wildlife / game

1) Is your land used for wildlife/game production?
Wildlife/game production means that animals are bought and/or sold, and/or supplementary feeding is provided e.g. ostrich, duiker, roan, sable, grey rhebuck and other non-domesticated animals.
☐ no, I do not buy "wild" animals to stock my property and I do not sell animals. I also do not provide supplementary feed to wild animals.
☐ yes, extensive farming (free ranging) for multiple purposes (e.g. ecotourism, animal sales and other)
☐ yes, semi-domestic intensive farming mostly for consumptive use but also other uses
☐ yes, both extensive farming and semi-domestic intensive wildlife farming

*If you **answered NO** above**, MOVE directly TO SECTION 4** and skip 3.1 and 3.2 below. If you **answered YES**, only complete the sections below that correspond to your answer above.*

Section 3.1: Extensive wildlife production

This section relates to free ranging non-domesticated wild animal production (whereby animals are bought and/or sold, and/or supplementary feeding is provided) for ecotourism, animal sales and other purposes.

1) What is the size of the area used for extensive wildlife/game production?
Approximate hectares or square kilometres (a percent/proportion of the total mountain property area can also be given if easier).

---

2) What are the main purposes?
MORE THAN ONE OPTION CAN BE SELECTED.
☐ ecotourism
☐ animal sales
☐ meat and other consumptive purposes
☐ private recreational hunting
☐ recreational hunting for a profit
☐ other:

3) Are the animals kept on the land all year round?
☐ no
☐ yes
☐ other:

4) What is the total number of wildlife used in this system on the property?
Estimated number of animals used in the extensive system at any time during the year.

---

5) What are the dominant types of wild animals stocked and managed?
Please include all information that you feel is relevant for the study to describe the wildlife production system used.

Section 3.2: Intensive wildlife production

This section relates to intensive semi-domesticated wild animal farming (e.g. ostrich, roan and others) mostly for consumptive use but also other uses.

1) What is the size of the area used for intensive wildlife production?

Approximate hectares or square kilometres (a percent/proportion of the total mountain property area can also be given if easier).

2) What is the total number of wildlife used in this system on the property?

Estimated number of animals used in the intensive system at any time during the year.

3) What type of wildlife is farmed in the intensive production system and for what purpose?

Please include all information that you feel is relevant for the study to describe the intensive wildlife production system used.

Section 4: "Wild" plant harvesting or collecting

The term "wild" plant harvesting or collecting refers to flowers or any plant material harvested or collected from the natural veld/vegetation.

***1) Is your land used for collecting or harvesting wildflowers or plant material?**

☐ no
☐ yes, occasionally
☐ yes, regularly
☐ other:

*If you **answered NO** to the question above**, MOVE directly TO SECTION 5** and skip 4.1 below.*

Section 4.1: "Wild" plant harvesting or collecting

***1) What type of "wild" plant material is harvested from the natural vegetation?**

☐ buchu
☐ some flowers and buchu
☐ flowers and other ornamental material
☐ other:

***2) What is the size of area used for wild plant harvesting/collecting on your property?**

Approximate hectares or square kilometres (a percent/proportion of the total mountain property area can also be given if easier)

3) If necessary, please provide more detail on the types and use of the wild plants and flowers harvested/collected.

Section 5: Cultivation

***1) Is your land used for cultivation?**

For example, annual or perennial crops, cut flowers, orchards, plantations or any other form

☐ no
☐ yes
☐ other:

*If you **answered NO** to the above, please **MOVE directly TO SECTION 6** and skip 5.1 below.*

Section 5.1: Cultivation

***1) What type of cultivation occurs on your land?** *MORE THAN ONE OPTION CAN BE SELECTED.*

☐ grains (e.g. wheat, beans)
☐ orchards (e.g. berries, nuts, citrus fruits, other tree fruit and grapes)
☐ flowers (e.g. protea cut flowers and other plant parts)
☐ herbs, oils or teas (e.g. buchu or rooibos)
☐ vegetables
☐ planted pastures
☐ plantations (timber)
☐ other:

***ONLY complete the SECTIONS directly** below that correspond to your answer to question 1 above.*

5.1.1 Grain crop questions

1) What portion of the area is used for grain crops?

Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the main types of grains planted?

Please name the most important or dominant grain types that are managed.

5.1.2 Orchard questions

1) What portion of the area is used for orchards?

Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the main orchards planted? *MORE THAN ONE OPTION CAN BE SELECTED.*

☐ berries
☐ nuts
☐ citrus fruits
☐ other tree fruits
☐ grapes
☐ other:

5.1.3 Cut flower questions

1) What portion of the area is used for cut flowers?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the principle cut flowers cultivated?
☐ Proteas (including *Protea*, *Leucospermum*, *Leucadendron* and other Proteaceae genera and species)
☐ other:

3) If necessary, provide any additional information about the cut flowers cultivated i.e. if specific types or species are used.

5.1.4 Herbs, oils or tea cultivation questions

1) What portion of the area is used for cultivating herbs, oils or tea crops?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the principle types of herbs, oils or tea crops cultivated?
Please name the most important or dominant types that are cultivated

5.1.5 Vegetable crop questions

1) What portion of the area is used for vegetable crops?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the principle vegetable crops cultivated?

5.1.6 Planted pastures questions

1) What portion of the area is used for planted pasture?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

5.1.7 Timber plantation questions

1) What portion of the area is used for timber plantations?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

2) What are the principle trees / species used?
Please indicate the most important or dominant trees / species that are used for the plantations.

5.1.8 Any other forms of cultivation

1) If applicable, please describe any other cultivation not covered above that occurs on your property.

2) What portion of the area is used for this cultivation?
Approximate hectares or square kilometres (a percent/proportion of the total farm area can also be given if easier)

Section 6: Recreation and tourism

***1) Is your land used for recreation or tourism?**
This includes both private (family and friends) and for profit or any other form

☐ no
☐ yes
☐ other:

Section 6.1: Recreation and tourism

***1) What type of recreation is your land used for?** *MORE THAN ONE OPTION CAN BE SELECTED*

☐ private (family and friends)
☐ facilities are provided for a profit (ecotourism)

☐ other:

2) What portion of your land is used for general recreational activities?

For example, hiking or other nature-based activities. Approximate hectares or square kilometres (a percent/proportion of the total property area can also be given if easier)

---

3) What portion of your land is used for recreational infrastructure and facility areas?

For example, chalets or camping areas. Approximate hectares or square kilometres (a percent/proportion of the total property area can also be given if easier).

---

4) What are the main activities / attractions? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ fynbos / cape floristic kingdom
☐ 4 x 4 routes
☐ mountain biking
☐ hiking
☐ wilderness experience
☐ camping
☐ accommodation other than camping
☐ swimming
☐ freshwater rivers for swimming
☐ fishing
☐ kloofing
☐ game viewing and/or drives
☐ hunting packages
☐ proximity to provincial nature reserve/wilderness area
☐ getting away from the city and relaxing/socialising in nature
☐ birding
☐ other:

5) What is the estimated number of visitors to your property per year for recreational purposes?

Section 7: Fire management

1) Do you burn your land?

I.e. do you use fire at all for any specific management reasons?

☐ no
☐ yes, occasionally
☐ yes, controlled block burning
☐ yes, patch burning [i.e. the approach used by some early farmers in the landscape prior to 1970]
☐ yes, indiscriminately
☐ I would like to burn but do not because of various reasons
☐ unknown
☐ other:

*If you **answered NO** to the above, please **MOVE directly TO SECTION 8** and skip 7.1 below.*

Section 7.1: Fire management

1) Why do you burn your land [or why would you like to burn your land]? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ for fire management, i.e. to reduce fire risk to economic and other activity
☐ to reduce fuel load to enable effective firefighting in the case of wild fires
☐ to stimulate new growth for forage for grazing livestock or wildlife
☐ to rejuvenate or thin the vegetation for recreation / tourism
☐ to stimulate the growth of specific plants, wildflowers or bulbs for recreation / tourism
☐ to stimulate the growth of specific plants, wildflowers or bulbs for harvesting
☐ to protect the natural vegetation i.e. loss of fire dependent plant types and species
☐ other:

Section 8: Mountain legislation and management

1) Have you heard of the Mountain Catchment Areas (MCA) Act of 1970?

The MCA Act of 1970 provides for the conservation, use, management and control of land situated in specific demarcated mountain catchment areas with the aim of protecting water supply. Land demarcated under the MCA Act of 1970 is recognized as a protected area in terms of the National Environmental Management: Protection Areas Act of 2003.

☐ no
☐ yes
☐ other:

2) Does the Mountain Catchment Areas Act influence your decisions as an owner and/or manager?

☐ no
☐ yes
☐ not really sure, because I am not familiar with the Act
☐ other:

3) Have you ever received any communication from any source related to the Mountain Catchment Areas Act?

☐ no
☐ yes
☐ other:

4) How do you feel about specific mountain areas being legislated under the Mountain Catchment Areas Act of 1970?

☐ neutral / non-committal
☐ positive (in favour)

☐ negative (strongly against)
☐ not really sure, because I am not familiar with the Act
☐ other:

5) Do you feel that there is merit in the state regulating activities in privately owned mountain areas to promote sustainable land use for conservation and for protecting water supply? *MORE THAN ONE OPTION CAN BE SELECTED.*

☐ no, the opportunity costs are too large (i.e. regulating the land reduces the economic viability and potential of the land)
☐ no, similar levels of current and future use benefits (e.g. recreation, water supply, education) can be provided from privately owned land without state intervention
☐ no, similar levels of protection for plants and animals (i.e. for intrinsic reasons, ethical and philosophical reasons) can be achieved on privately owned land without state intervention
☐ yes, state regulation can assist for protecting and providing current and future use benefits (e.g. for recreation, water supply, education among other uses)
☐ yes, state regulation can assist for protecting plants and animals (i.e. for intrinsic reasons, ethical and philosophical reasons)
☐ yes, but with effective communication and collaboration
☐ yes, as long as it does not affect the economic viability of the land
☐ no, no real comment
☐ not really sure
☐ other:

6) Would you suggest that the Mountain Catchment Areas Act be revoked?

☐ no
☐ yes
☐ not really sure, because I am not familiar with the Act
☐ other:

7) Do you know if your property falls under the Mountain Catchments Areas Act of 1970?

☐ I am not sure, because I am not familiar with the Act
☐ my property does not fall under land demarcated by the MCA Act
☐ my property does fall under land demarcated by the MCA Act
☐ other:

8) If necessary, please use the space below to explain or provide further details to your answers above.

Section 9: Fire legislation and management

1) Are you aware of the National Veld and Forest Fire Act of 1998?

The NVFF Act places certain responsibilities on the landowner from whose properties a fire may start and spread.

☐ no
☐ yes
☐ not really sure
☐ other:

2) Does the National Veld and Forest Fire Act influence your decisions as an owner and/or manager?

This question is based on a linear scale from 1 to 5 whereby 1 = yes extremely influential, 2 = influential, 3 = moderately influential, 4 = not really influential, and 5 = Not at all.

Yes, extremely influential	1 ☐	2 ☐	3 ☐	4 ☐	5 ☐	**No not at all**

3) What are your views on the types of fires currently affecting the mountain catchment? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ extensive
☐ highly destructive
☐ uncontrollable
☐ small
☐ not a problem
☐ controllable
☐ burning for extended periods >7 days
☐ burning for many days (e.g. 5-7 days)
☐ burning for 2-3 days
☐ burning for 1 day
☐ no views really
☐ other:

4) What do you think is the ideal way to regulate and manage fire in the mountain catchment? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ exactly how it is being regulated currently i.e. through the National Veld and Forest Act of 1998
☐ exactly how it is being regulated and managed i.e. through the National Veld and Forest Act of 1998 with support from Fire Protection Associations
☐ a revised version of the current system
☐ a revised version of the current system including elements of historical management approaches
☐ a completely new approach
☐ other:

5) What are your views of the patch burning system used by farmers during the early to mid-20th century?

Some historical records indicate that early farmers used to go through the mountain veld annually in summer and ignite all patches of veld that would burn to provide pasture for stock. Not all patches would burn and this would create a mosaic pattern in the landscape of different veld ages. In certain cases, buchu-growing areas would also be burnt every two years in the mountains to stimulate growth.

☐ I am not aware of this system
☐ no real views on this system
☐ it was a relatively effective system for managing fire in the landscape
☐ it was not effective for managing fire in the landscape
☐ certain elements of the system could be useful for managing fire in the landscape
☐ completely against it

☐ other:

6) What are your views of controlled block burning conducted on a rotational basis to reduce fire risk and fuel accumulation?

☐ I am not familiar with this as a management option
☐ no real views
☐ if managed correctly, it is a relatively effective system for managing fire in the landscape
☐ it is not effective for managing fire in the landscape
☐ completely against it
☐ other:

7) If necessary, please use the space to elaborate and provide additional detail to your answers above.

Section 10: Alien (invader) plants

1) Are you actively managing aliens on your property?

☐ no, it is too resource intensive
☐ no, no real comment
☐ yes, I make an effort
☐ yes, I make a substantive effort
☐ I do not know – not really sure
☐ other:

2) How would you describe the density of alien invader plants on your property?

Closed = the highest density, very scattered = the lowest density. The other options are in linear order from closed to very scattered.

☐ closed (highest density)
☐ dense
☐ moderate
☐ occasional
☐ rare
☐ scattered
☐ very scattered (lowest density)
☐ there are no alien invader plants on my property (no alien plants)
☐ I do not know – not really sure
☐ other:

*If you **answered, "I don't know" or "no alien plants" to question 2** above, **MOVE directly TO SECTION 11** and skip 10.1 below.*

Section 10.1 Alien (invader) plants

1) What is the main age of the alien invader plants on your property?

☐ mature
☐ young
☐ other:

2) What are the dominant types of alien plants? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ not sure
☐ wattles (black wattle, long leaf wattle, or silver wattle)
☐ eucalypts
☐ hakea
☐ sesbania
☐ port jackson
☐ pines
☐ poplars
☐ rooikrans
☐ other:

3) Where are invasions mostly in the landscape? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ riparian and wetland areas
☐ upland areas / koppies
☐ old fields
☐ lowland / flat areas
☐ other:

Section 11: Desired land use and management

1) Do you feel that your property is being used and managed in an economically effective manner?

☐ no
☐ yes
☐ partly
☐ economic effectiveness is not a goal for the property
☐ other:

2) What is stopping you from implementing changes to increase economic effectiveness? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ financial
☐ legislation
☐ climate variability
☐ climate change
☐ nothing – not relevant
☐ other:

3) Do you feel that your property is being used and managed in an environmentally sustainable manner?

☐ no
☐ yes
☐ partly
☐ environmental sustainability is not a goal for the property
☐ other:

4) What is stopping you from implementing changes to increase sustainability? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ financial
☐ legislation
☐ climate variability
☐ climate change
☐ nothing – not relevant
☐ other:

5) What management aspect would you change if you were provided the opportunity? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ improve fire use and management
☐ improve alien invasive management
☐ improve land use (e.g. change what you use the land for)
☐ improve my access to climate variability information
☐ improve my access to climate change information
☐ improve conservation related activities on my property
☐ improve visitor levels
☐ nothing – not relevant
☐ other:

6) If necessary, please provide further details of your ideal land use and management of your property.

Section 12: Wilderness Area

1) Do you know of the Groot Winterhoek Wilderness Area?

The "Wilderness Area" is a protected area managed by CapeNature situated in the southern section of the Groot Winterhoek Mountains. It includes the headwaters of the Twenty-four rivers. The area is fully protected from land use and all use is regulated and limited to recreation and tourism activities (NEM:PA Act of 2003).

☐ no
☐ yes
☐ other:

2) Have you visited the Groot Winterhoek Wilderness Area? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ no
☐ yes, for recreation
☐ yes, for other reasons

If other reasons, please describe reasons for visiting:

3) Are you aware that certain properties in the Wilderness Area were expropriated during the 1960-80s to ensure the continued supply of clean water for various uses?

Expropriation includes monetary compensation for land owners affected. Certain properties were voluntary sold.

☐ no
☐ yes
☐ other:

4) What are your views on the expropriation of properties in the past for the creation of the Wilderness Area and for the protection of water sources?

☐ neutral / non-committal
☐ positive (in favour)
☐ negative (strongly disagree)
☐ other:

5) Do you feel that there is merit in the Wilderness Area being fully protected by the state as opposed to privately owned and managed? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ no, the opportunity costs are too large (i.e. the land could be used for something more economically viable)
☐ no, similar current and future use benefits could be provided from privately owned land in the same area (e.g. for recreation, water supply, education)
☐ no, similar levels of protection for plants and animals could be achieved on privately owned land in the same area (i.e. for intrinsic, ethical and philosophical reasons)
☐ no, the wilderness area does not influence land owner decisions or management practices in surrounding and nearby properties
☐ yes, because the wilderness area provides protection for current and future use benefits (e.g. for recreation, water supply, education among other uses)
☐ yes, because the wilderness area provides protection for plants and animals for intrinsic, ethical and philosophical reasons
☐ yes, but greater collaboration is required with other landowners and other interested individuals
☐ yes, but there should be more incentive to generate revenue and to increase management effectiveness and accessibility
☐ no, no real comment
☐ not relevant, I am not familiar with the Wilderness Area
☐ other:

6) Does the Wilderness Area influence your property at all? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ not relevant, I am not familiar with the Wilderness Area
☐ no, no real comment
☐ no, there are no meaningful links between my property and the wilderness area
☐ yes, it encourages a sustainability ethos in my land use and management
☐ yes, it partially influences my land use and management decisions towards a more sustainable manner
☐ yes, it strengthens the tourism and recreational appeal of my property
☐ other:

7) Do you value the Wilderness Area and for what? *MORE THAN ONE OPTION CAN BE SELECTED*

☐ not relevant, I am not familiar with the Wilderness Area
☐ no, no real comment
☐ yes, for hiking, relaxing and recreation (direct use)
☐ yes, for spiritual or cultural experiences/reasons (direct use)
☐ yes, for continued water supply (indirect use)
☐ yes, for soil erosion control (indirect use)
☐ yes, for future generations to experience and utilise (non-use value)
☐ yes, I feel good knowing that the wilderness area is there for protecting animals, plants and ecosystems (non-use value)

☐ yes, plants, animals and ecosystems have a safe place to exist in the Wilderness Area (intrinsic, ethical and philosophical reasons)

☐ I do not know – not really sure

☐ other:

8) Have you heard of the "Greater Cederberg Biodiversity Corridor" or the "Groot Winterhoek Freshwater Stewardship Corridor"?

MORE THAN ONE OPTION CAN BE SELECTED

☐ no

☐ yes, the Greater Cederberg Biodiversity Corridor

☐ yes, the Groot Winterhoek Freshwater Stewardship Corridor

☐ other:

9) Is your property or parts thereof under any voluntary conservation stewardship agreements?

The Biodiversity Stewardship Programme is an entirely voluntary conservation programme run by CapeNature in the Western Cape which offers four voluntary stewardship agreements to landowners: i) Nature Reserve, ii) Biodiversity Agreement, iii) Protected Environment, and iv) Conservation Area. Each has its own incentives (benefits) and restrictions. *MORE THAN ONE OPTION CAN BE SELECTED*

☐ no, I am not aware of this programme

☐ no, but I am aware of the programme

☐ yes, voluntary conservation area (flexible option including registration with CapeNature)

☐ yes, biodiversity agreement (contractual biodiversity agreement between CapeNature and landowner)

☐ yes, protected environment (includes declaration under Protected Areas Act (S28 of NEM:PA Act))

☐ yes, nature reserve (includes declaration under Protected Areas Act (S23 of NEM:PA Act))

☐ other:

*If you **answered** "**NO**, I am not aware of this programme or **NO**, but I am aware of the programme", **MOVE directly TO SECTION 13** and skip question 9 below.*

10) What portion of your land falls under the agreement indicated above?

Approximate hectares or square kilometres (a percent/proportion of the total property area can also be given if easier)

Section 13: Past land use

1) Please provide any information that is relevant to the past land use, alien plant coverage or fire use or management of your property.

This can include for any dates or times between 1900 and the current day.

a) Past land use (include dates if possible)

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b) Past fire use or management (include dates if possible)

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c) Past alien plant coverage (include dates if possible)

Section 14: Change in the value of the mountain catchment

This section is about how you perceive change (if any) to have happened in the mountain catchment and how you feel about this change.

Section 14.1: Economic value and Environmental sustainability

1) Please list and describe any significant events that you are aware of that have resulted in changes (positive or negative) to the <u>economic</u> value or the <u>environmental sustainability</u> of the catchment.

This can include any short-term or long-term disturbances or positive events that have occurred over the last decade or as far back as you can remember that has influenced your property or other property owners to change how they use or manage the land or even influenced landowners in the catchment in any way.

Appendix 5.1 A semi-automatic aerial triangulation approach to the orthorectification of historical aerial images covering mountainous terrain and constrained by film deformations

ABSTRACT

A major constraint to upscaling the use of historical aerial imagery for 20th century change detection is challenges associated with accurate orthorectification. Geometric errors associated with complex and mountainous terrain, lens distortions and film and print shrinkage can complicate the spatial referencing process. Furthermore, establishing a reliable ground control network (e.g. 4-20 ground control points per image for single image rectification and geo-referencing) is a time consuming and cumbersome process. Automatic aerial triangulation with bundle block adjustment provides an opportunity to automate the process and reduce the number of ground control required, whilst working with larger blocks of images. However, the feasibility of this approach has not been fully explored for historical aerial imagery. This study used industry standard photogrammetric and orthorectification software to test the potential and accuracy of digital automatic aerial triangulation with bundle block adjustment and reduced ground control for orthorectifying historical aerial images covering mountainous terrain and containing deformations related to film warpage and shrinkage. Specifically, the effects of different aerial triangulation post-processing models were tested on the final planimetric accuracy of orthoimages generated considering photo jobs with a range of photo numbers (12-237 images) and area coverage (>33 000-~150 000 ha). Adjustments were made to blunder removal, self-calibration and ground control standard deviation settings in the block bundle adjustment processing. Although the aerial triangulation process could not be fully automated the number of ground control points was substantially reduced from the recommended 4-20 per image to approximately one point every sixth image with acceptable final planimetric accuracy being achieved for certain post-processing models. Self-calibration with 44 parameters and fixing the model to the ground control network greatly improved final planimetric accuracy (total root mean square error [RMSE] ranging from 18.6 – 25.7px [7.1 – 21.8m] at ground control points and 20.5 – 27.7px [7.8 – 23.5m] at checkpoints). Removing automated points and running post-processing with no self-calibration increased the error at checkpoint locations (total RMSE ranging from 24.1 – 33.5px [9.2 - 28.4m] with maximum RMSE increasing by >47px [30m]). Allowing movement in the model by increasing standard deviations at ground control points or automating the removal of blunders in the model significantly reduced final planimetric accuracy. Aerial triangulation accuracy results were not an accurate reflection of the error in the final orthoimages. This was substantiated by limited and negative correlations between aerial triangulation overall accuracy and final orthoimage accuracy. Further work should focus on attempting to increase final planimetric accuracy by adjusting the accuracy and number of

manual tie points and ground control in combination to altering the amount and positioning of automatic tie points.

1. Introduction

With the advancement of satellite technology over the last four decades, high-resolution aerial and satellite images (e.g. 0.82m from IKONOS to 0.3m WorldView images) are increasingly being used to evaluate and monitor a wide array of environmental and climatic variables (Bianchetti & MacEachren 2015; Chen et al. 2016). Despite recent progress, historical temporal and spatial coverage and poor resolution limits the use of satellite images for determining landscape dynamics during the 20th century i.e. >20-80 years ago (Gennaretti et al. 2011). For example, satellite imagery is only available from the 1970s and the resolution available from Landsat1 is 80m (for the 1970-80s) and from Landsat7 is 30m (from the 1990s) (Chen et al. 2016). Additional errors in sensors and cloud cover reduce the applicability of historical satellite imagery for determining fine-scale and longer term 20th century environmental change.

Historical analogue aerial photos provide a unique opportunity to document the past 20-80 years of landscape change. Photos date back to the early twentieth century and are available from many governmental and certain private institutions across the globe (Tekle & Hedlund 2000; Palandro et al. 2003; Schiefer & Gilbert 2007; Gennaretti et al. 2011; Ma & Buchwald 2012; Abrate et al. 2013; Asiyanbola 2014). If consolidated, through country specific research programmes, historical aerial imagery could provide an opportunity to construct a global picture of 20th century change, and a baseline for comparisons to current and future satellite and airborne imagery of the 21st century (Ma & Buchwald 2012).

A major constraint to upscaling the use of historical aerial photos is challenges associated with accurately spatially referencing a large set of historical aerial images for them to be overlaid and/or quantitatively compared to other geographic datasets including satellite imagery as well as current orthoimages. Challenges are mostly linked to the type and distribution (systematic or random) of geometric errors (distortions and displacements) present in the imagery and the time-consuming and cumbersome process of collecting a reliable network of ground control information. Geometric errors influence the amount of manual effort required to spatially reference an aerial image as well as influence the final accuracy of the location and size of features within a landscape. For example, geometric errors can result in certain cover classes being overestimated by up to double the occupied area during classification and change detection studies (Rocchini 2004; Wang & Ellis 2005a; Morgan et al. 2010; Nagarajan & Schenk 2016).

It is well described in the literature that topographically complex and highly mountainous areas present greater challenges for accurate spatial referencing than areas of flat and simple terrain (Rocchini & Di Rita 2005; Wang & Ellis 2005a, 2005b; Rocchini et al. 2012). In flat areas, it has been proven that polynomial functions for rectification may be acceptable for use. However, when polynomial functions are used in rugged terrains they show a substantial increase in planimetric error. This is because polynomial rectification cannot correct for relief displacement, as there is no information on elevation used in the rectification algorithms. Orthorectification, on the other hand, results in much improved accuracy in comparison to polynomial functions for all types of terrain. This is mainly due to the orthorectification method taking into account the elevation of the area under study by means of a digital elevation model (Rocchini & Di Rita 2005; Rocchini et al. 2012).

Film or print shrinkage or warpage is another cause of geometric distortion and particularly of relevance to historic photographs and film (Morgan et al. 2010). Deformations linked to historic film or print shrinkage are dependent on a range of factors including the image constitution i.e. type of material of the negative photo, past country context and circumstances and changes in management and storage facilities. With the increase in digital acquisition of aerial images and methods, photogrammetric equipment (such as high quality photogrammetric scanners) are not always available in-house where analogue aerial photos are stored. This is especially the case in developing and least developed countries where understaffing and budget constraints are common and priorities for land development departments are justifiably aimed at provision of settlement services and housing to local communities (Asiyanbola 2014). Errors associated with film deformations are independent to terrain roughness errors however; these two sources of error combine in complex ways to have much larger impacts on planimetric accuracy (Morgan et al. 2010).

Current literature and country initiatives aimed at spatially referencing and geometrically correcting historical aerial images for change detection have mainly been driven through the standard procedures of manually locating an extensive number of ground control points that can also be located on an existing orthoimage or spatially referenced satellite image. This includes approximately 4-20 ground control points per photo (or photo pair) or approximately 30- 60 (or more) ground control points for approximately 10 000ha (Hughes et al. 2006; Rocchini et al. 2006; Marignani et al. 2008; De Rose & Basher 2011; Gennaretti et al. 2011; Abrate et al. 2013; Pulighe & Fava 2013). These coordinates are then used to transform the image to the ground control coordinate system using polynomial rectification or standard photogrammetric aerial triangulation approaches for orthorectification. A major hindrance in this process is the location of a large number of ground control points for every photo across all years and photo jobs (Nagarajan & Schenk 2016). This approach may seem feasible for small areas or for photo jobs comprising a few photos (Hervás et al. 2003; Ellis et al. 2006; De Rose & Basher 2011), however when considering large photo jobs and study areas, e.g. greater than 200 photos and covering an area greater 150 000 ha (1500 km^2), it can become an arduous process. For example, finding the same object in the early 1900s and then again in current imagery can be particularly problematic especially when working in natural, remote and mountainous regions (Wang & Ellis 2005b; Hughes et al. 2006; Ma & Buchwald 2012; Nagarajan & Schenk 2016).

Over the last decade, automatic aerial triangulation has advanced the efficiency of digital photogrammetry to a new level. This is mainly through the development of automatic image orientation which reduces time-consuming block preparation and interactive manual input (Krzystek et al. 1996; Schenk 1996; Chen et al. 2016). Automatic image orientation aims at reconstructing the coordinates of the perspective centre, the viewing direction during image capture and the interior orientation parameters of a potentially very large set of unordered or ordered images (Chen et al. 2016). Several steps are required during the process, all mostly aimed at determining the exterior orientation (EO) of each image. The steps include locating features (known as tie points) in overlapping images as well as ground control points (XYZ - extracted from a reference layer) in multiple images and then using this information to perform a bundle block adjustment computation. The bundle block adjustment is used to calculate the exterior orientation (EO) for each image from a block of a few images and even up to a couple thousand images. The method essentially adjusts all the images together until they make a connected block and then adjusts the whole block until it fits the ground control points. The main goal being to statistically find the best possible fit for each image to each other but also for the block of images to fit the ground (based on the ground control points) (Schenk 1996; Linder 2006; Abrate et al. 2013; Chen et al. 2016; Nagarajan & Schenk 2016).

Automatic aerial triangulation with bundle block adjustment potentially provides an opportunity to increase the automation of orthorectification of historical aerial imagery and to reduce the amount of ground control required across a large set of aerial images (Linder 2006; Addo 2010). This is through automatic tie point image matching using feature based and least square matching techniques, bundle block adjustment, and blunder removal (Schenk 1996; Addo 2010; Verhoeven et al. 2012; Abrate et al. 2013; Chen et al. 2016). However, the advantages of these methods (including using a reduced number of ground control points) have not been fully explored for spatially referencing and geometrically correcting historical aerial images.

This paper presents an assessment of the potential for using digital automatic aerial triangulation with bundle block adjustment for orthorectifying damaged and degraded historical aerial images covering 30 000 – 150 000 hectares of mountainous and rugged terrain. The main aim of the paper is to describe the effects of different aerial triangulation post-processing models on final planimetric orthorectification accuracy with a reduced ground control network. The paper intends to contribute to the global discussions on the current methodology for extracting and analysing spatiotemporal information of objects from historical aerial images and image sequences by using approaches from photogrammetry with emphasis on accurate and reliable geometric information but with reduced effort in terms of the number of ground control points required. The paper focuses on the following key questions:

1) What level of accuracy can be achieved for damaged and degraded photos covering a large mountainous area using digital automatic aerial triangulation and bundle block adjustment with reduced number of ground control points?

2) What level of automation can be achieved for an acceptable level of accuracy?

3) What contradictions exist between aerial triangulation root mean square error results and final planimetric accuracy in orthoimages generated?

4) Can consistencies be found in terms the post-processing model applied and final planimetric accuracy across different photo jobs? Photo jobs differ in terms of the flight year, number of photos, scale and area covered i.e. is there a consistent approach that can be used for increasing orthoimage planimetric accuracy?

5) How do results compare to other aerial image georeferencing and registering approaches that have been used in past studies investigating landscape change?

2. A summary of approaches to spatial referencing and geometric correction

There have been several methods proposed for spatially referencing and geometrically correcting historical aerial photos. A review of methods for digital imagery is presented in Novak (1992) and a more recent literature review is presented in Nagarajan & Schenk (2016). Studies generally refer to georeferencing or the registration of aerial images interchangeably but also as an essential photogrammetric component of the orthorectification process. The two terms are also used to describe polynomial and projective transformation approaches among others (Hughes et al. 2006; Abrate et al. 2013). The following aspects have been researched and established in the change detection literature:

- Georeferencing based on photogrammetric approaches (also referred to as registration of aerial images by exterior orientation) is essential for orthorectifying aerial images (Schenk 1996; Nagarajan & Schenk 2016).
- Georeferencing using polynomial and projective models is fundamentally a two dimensional transformation based on x and y ground coordinates whereas orthorectification is a three dimensional transformation which accounts for the height i.e. based on x, y and z coordinates (Rocchini et al. 2012).
- Positional accuracy can be critical for the classification processes and small errors can significantly affect the size and location of features in the landscape (Rocchini 2004; Wang & Ellis 2005a; Rocchini et al. 2006, 2012).
- Positional accuracy is influenced by geometric errors, which are caused by random or systematic distortions and displacements (Morgan et al. 2010).
    - Systematic error such as lens distortion (more common on old photographs) and earth curvature
    - Random error such as detector error (roll, crab/yaw, pitch), film or print shrinkage (especially relevant for historic photographs or film) and atmospheric refraction of light
    - Topographic/relief displacement is a form of random geometric error and is more obvious in mountainous areas.

- Distortions caused by image motion compensation can cause systematic geometric errors and typically occurs on high-resolution aerial images.

- Orthorectification increases the planimetric accuracy of aerial photos in comparison to polynomial and projective transformations and especially in areas with mountainous and topographically complex terrain (Rocchini et al. 2012; Nagarajan & Schenk 2016).
- Depending on the application of a study, polynomial and projective model approaches can be considered but only when working in flat terrain. There can be as much as an increase of 60-160m of error when polynomial functions are used in rugged terrains, therefore being inappropriate for landscape change detection purposes when uncertainty on terrain roughness occurs (Rocchini & Di Rita 2005; Rocchini et al. 2012).
- Generally, there are two imaging geometry models used in orthorectification: rigorous camera models and rational function models (Kaichang et al. 2003; Ma & Buchwald 2012).
- In order to use the rigorous camera model to represent the imaging geometry, physical parameters about the camera are required including focal length, principle point location, pixel size, and lens distortions and orientation parameters (Ma & Buchwald 2012).
- Collinearity conditions are the most popular equations used to implement the transformations based on the rigorous model however; others can also be used and are available (Ma & Buchwald 2012; Nagarajan & Schenk 2016).
- Rational functions have been applied in photogrammetry to represent the transformation between the image space and the object space whenever rigorous models are unavailable due to limitations in information and parameters (Kaichang et al. 2003).
- Rational function models, however, are generally only appropriate for small areas with gentle terrain and require a large number of ground control points (e.g. 0.4 gcps km^{-2}). This is in particular for mountainous areas (Wang & Ellis 2005b).
- For stable and accurate results, exterior orientation and self-calibration by rigorous photogrammetric methods are recommended (Nagarajan & Schenk 2016).
- Regardless of the method used, a reliable network of ground control information is essential, a process that can be extremely time consuming (Schenk 1996; Wang & Ellis 2005b; Nagarajan & Schenk 2016).
- Ground control points can be estimated using recent maps, orthoimages, digital elevation models and field surveys all of which have respective pros and cons related to accuracy and manual effort (Wang & Ellis 2005b; Nagarajan & Schenk 2016).
- Studies generally recommend and use on average 4-20 ground control points per image or image pair depending on the georeferencing or registration process being used (Hughes et al. 2006; Rocchini et al. 2006; Marignani et al. 2008; De Rose & Basher 2011; Gennaretti et al. 2011; Abrate et al. 2013; Pulighe & Fava 2013).
- Automatic aerial triangulation with bundle block adjustment provides an opportunity to reduce the number of ground control required (Schenk 1996; Linder 2006). A minimum of three ground control points are required for a block of images, comprising few to several thousand, with best practice being to include one ground control point in every 6^{th} image

(3rd image pair) along the borders of the block with additional height control points inside the block (Linder 2006).

- Certain studies using bundle block adjustment still include relatively extensive ground control networks e.g. 30-60 ground control points for areas of 10 000 ha and 5 to 26 photos (Wang & Ellis 2005b).
- Georeferencing accuracy using polynomial and projective transformations is verified through the root mean square error calculation of ground control positioning accuracy (Rocchini et al. 2012).
- The root mean square error is also used to determine the accuracy of the different steps conducted in the photogrammetric process including the residuals in the ground control points but also for the manual and automatic tie points between images (Linder 2006).
- Final orthorectification error is determined after orthorectification by means of a digital terrain/elevation model and is achieved by comparing positions on the final historical orthoimages to the reference image from which the ground control points were extracted (Wang & Ellis 2005b; Schiefer & Gilbert 2007).
- When reviewing the accuracy of georeferencing and orthorectification the root mean square error of checkpoints in addition to ground control points should be used. Checkpoints are not included within the model translation and therefore is a way to double check the accuracy across the entire photo surface i.e. not only measured at ground control points (Schiefer & Gilbert 2007; Abrate et al. 2013).
- Studies are showing promising results for using ground control features in addition to points (i.e. lines and polygons as opposed to points) (Schenk 2004; Nagarajan & Schenk 2016).

3. Materials and Methods

3.1 Study area

Automatic aerial triangulation approaches and parameters were tested for a mountainous area in the Western Cape of South Africa (Figure 1) using historical aerial photos sourced from South Africa's national mapping organisation, the National Geospatial Information (NGI) office, a Chief Directorate of the Department of Rural Development and Land Reform (DRDLR).

South Africa provides an ideal case study for this work as the NGI has an extensive collection of aerial photos with coverage across the country dating from the early 1900s. Most of these have not been rectified potentially because of limited awareness of the need and usage of such historical imagery as well as certain technological, financial and institutional barriers (Figure 2 and Figure 3).

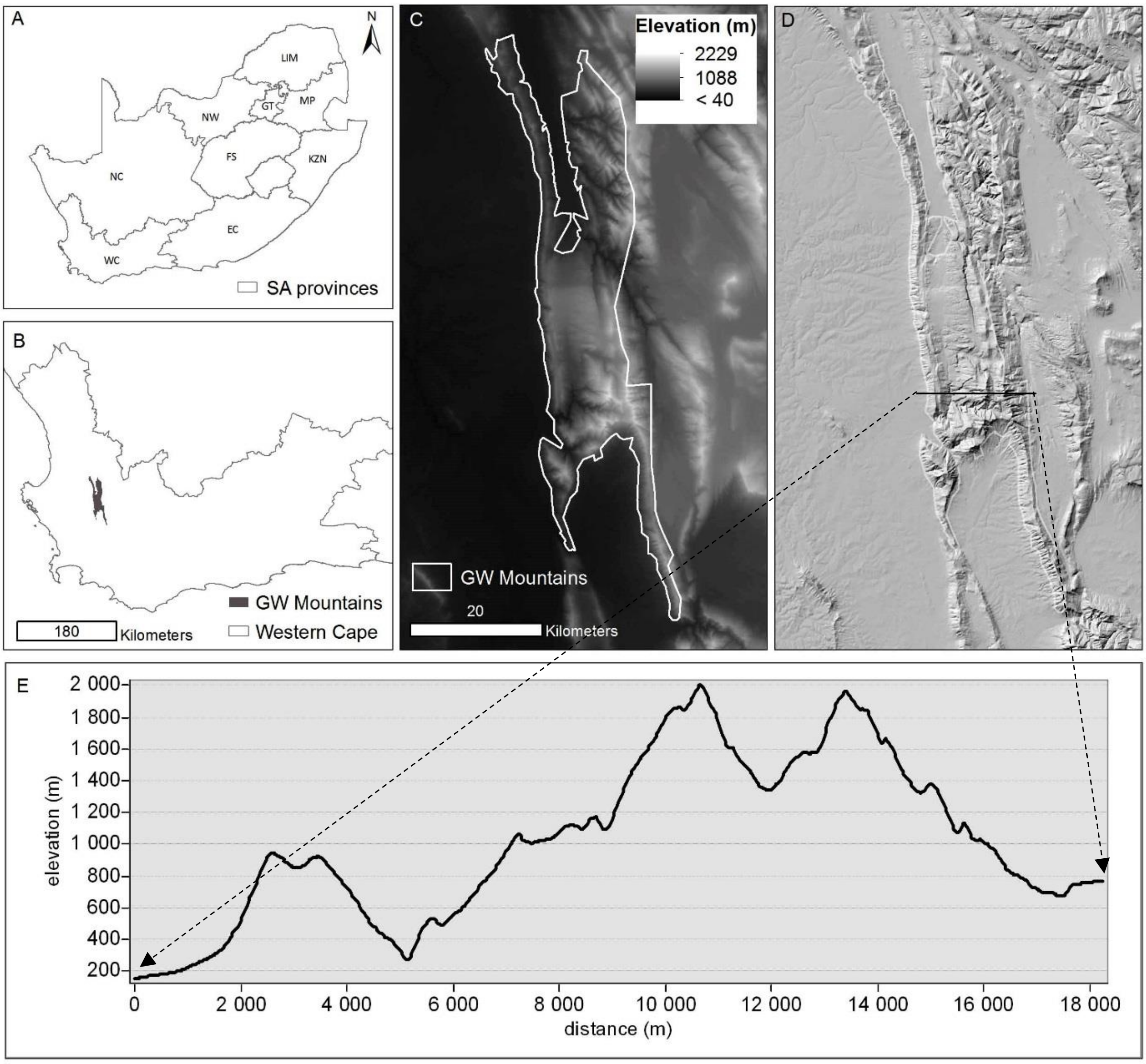


Figure 1 Study area: the Groot Winterhoek Mountain Catchment (highest elevation reaching 2077m) in the Western Cape of South Africa (A-E). Elevation and hillshade are shown in C and D and an elevation profile for one cross section of the catchment is plotted in E as an indication of the complexity of the topography. South African provinces: Western Cape (WC); Eastern Cape (EC); Northern Cape (NC); North West (NW); Free State (FS); Kwazulu Natal (KZN); Limpopo (LIM); Mpumalanga (MP); Gauteng (GT) (MDB 2013; SUDEM 2014; DEA 2016; NGI-DEM 2016)

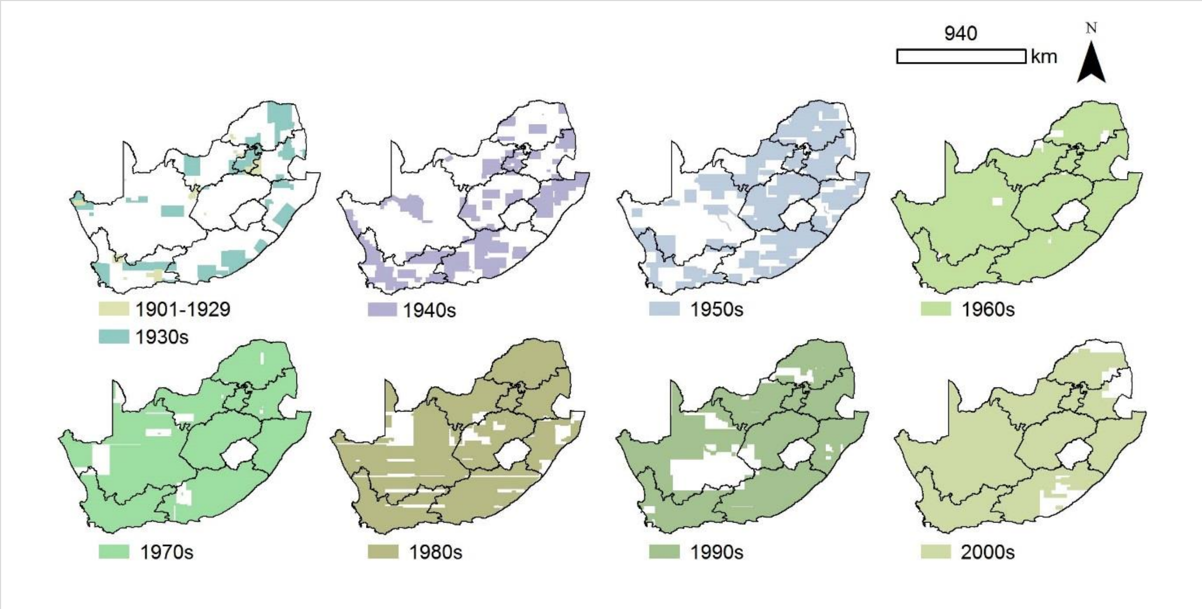


Figure 2 National coverage of non-rectified aerial imagery available for South Africa from the NGI with scales ranging from 1:5000 and 1:50 000 (original data source: NGI (2015a, 2015b)).

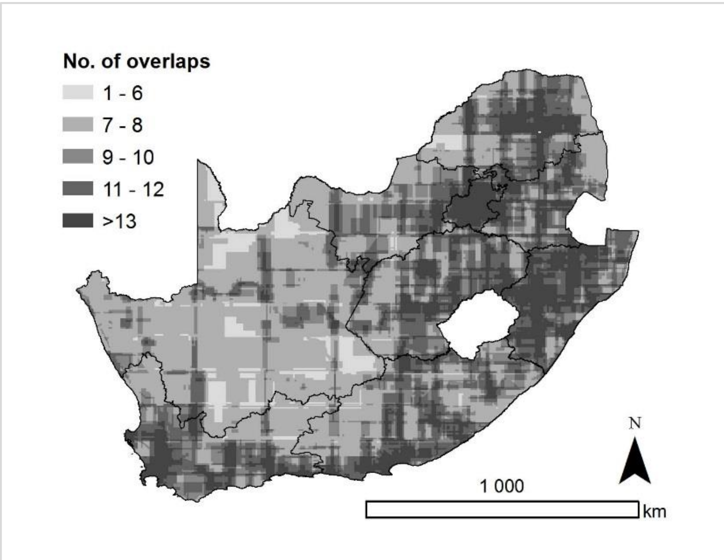


Figure 3 The number of overlapping years of non-rectified aerial photos available from NGI with scales ranging from 1:5000 to 1: 50 000 (original data source: NGI (2015a, 2015b)).

The focus was to find full photo coverage of the study area (the Groot Winterhoek Mountains) representative of the earliest available record of aerial photos and then a mid-time period between the earliest and current dates. Eleven photo jobs were available for the Groot Winterhoek Mountains i.e. photo jobs that were found to cover parts of the catchment. Four photo jobs were prioritised based on flight year, photo scale and time period. These photos together covered the full catchment but also extended into the surrounding lowland fringes and incised valleys. The photo jobs were representative of the late 1940s and early 1970s (Figure 4).

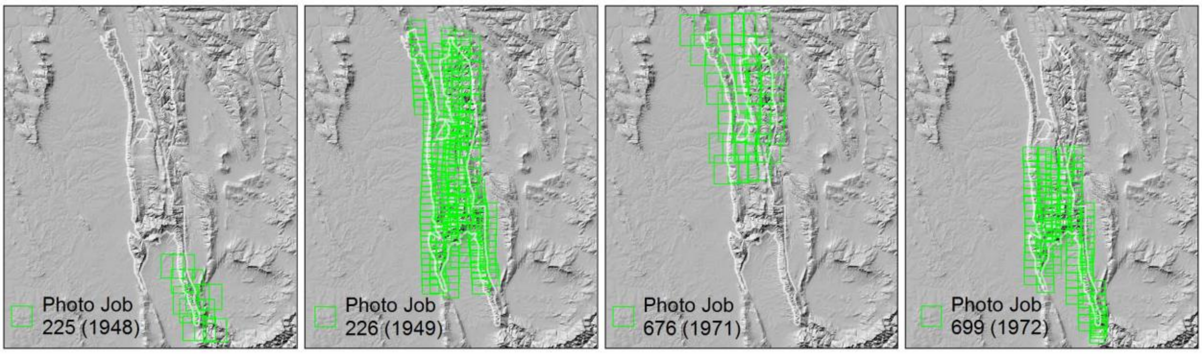


Figure 4 Footprints and boundaries shown based on the orthoimages generated in the study to indicate the overall photo job coverages (orthoimages generated using Trimble/Inpho 2014, SUDEM 2015).

3.2 Orthorectification and accuracy assessment

The study comprised four main components detailed in Figure 5 and further described in Section 4.2.1 to 4.2.4. Aerial triangulation in the form of bundle block adjustment and orthorectification was conducted respectively using MATCH-AT and OrthoMaster within Applications Master from Inpho (Trimble/Inpho 2014a). The final accuracy assessment of the orthoimages generated was manually determined using ArcGIS software (ESRI 2015). R Studio was used for data analyses, correlations and graphs (R Core Team 2016).

3.2.1 Input: historical and reference data, images and DEM

High-resolution scans of historical aerial photo negatives (21 microns per pixel) were obtained from South Africa's National Geospatial Information (NGI) office for 1948/9 and 1971/2. Camera details including radial distortions, fiducial mark coordinates and calibrated focal lengths were extracted from historical flight plans and camera calibration reports stored at NGI (Table 1). The analogue film negatives were warped to different degrees (Figure 6). These film deformations were also visually evident in the digital images after the scanning process.

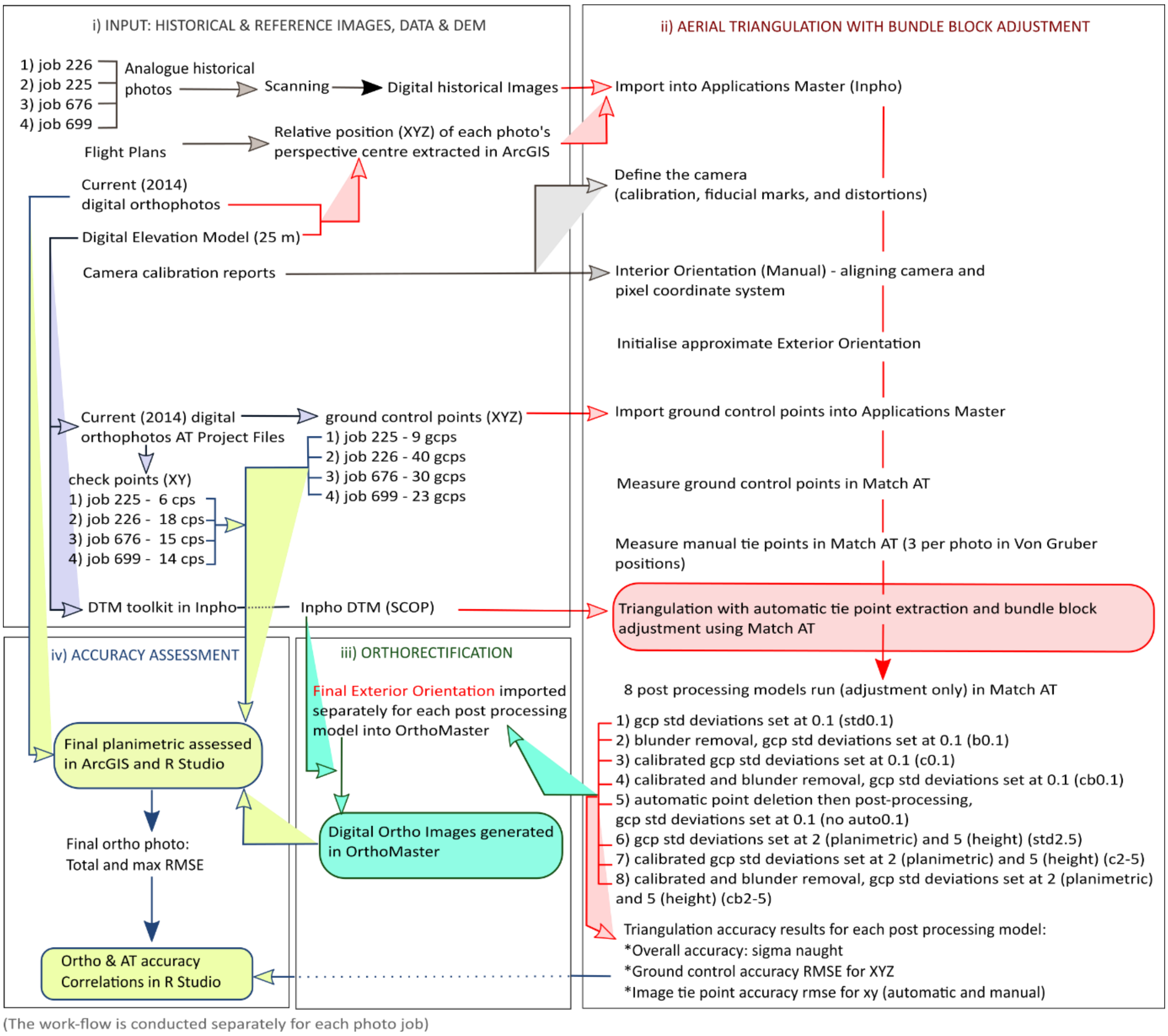


Figure 5 Study workflow consisting of four main components i) input images, data and DEM; ii) aerial triangulation with bundle block adjustment conducted in Inpho MATCH-AT photogrametric software; iii) orthorectification conducted in Inpho OrthoMaster software; iv) accuracy assessment conducted in ArcGIS and R Studio.

Table 1 Historical film based aerial photos including relevant scales and dates flown.

Job no.	Date flown	Scale	Area (ha)	No. of images (strips)	Focal length (mm)	Image size (cm)	No. of gcps/cps
225	Dec 1948	1:30 000	33 155	12 (5)	152.986	22.86*22.86	9/6
226	Jan-Feb 1949	1:18 000	156 800	237 (8)	152.986	22.86*22.86	40/18
676	Jan 1971	1:40 000	116 652	38 (8)	152.5	23*23	30/15
699	Feb 1972	1: 20 000	95 370	130 (7)	152.55	23*23	23/14

Figure 6 Examples of deformations on original photo negatives stored at NGI and image container (Photos: P Holden).

Current 2013/2014 digital orthoimages were also obtained from NGI including the respective aerial triangulation project files (Intergraph project files) and a photogrammetrically compiled 25 m Digital Elevation Model (DEM) (absolute accuracy 2.5 – 5m) for use in the orthorectification process.

The approximate relative position (XYZ) of each historical image (i.e. perspective centre of each image) was determined in ArcGIS (ESRI 2015) using a digitised version of the flight plan for each photo job and the 25m DEM. Easting and Northing coordinates were extracted from the digitised flight plan based on the current set of orthoimages. Average terrain height was extracted using the photogrammetrically compiled DEM. The average flying heights (Z) (i.e. altitudes of the average photo perspective centres) were then calculated by multiplying the focal length by the photo scale and adding this to the average terrain height.

Ground control points (XYZ) were measured directly from the current (2013/4) orthoimages' aerial triangulation Intergraph project files. At least one ground control point (gcp) was located for every sixth to eighth historical image per flight strip. This was initially conducted across all four-photo jobs. The number of ground control points, however, was increased for job 676 from 19 to 30 gcps and job 225 from 7 to 9 gcps (Figure 5). This was conducted as a function of the area covered by these photo jobs in relation to the area and number of gcps used for the largest photo job 226 (in terms of area coverage and number of photos). For job 226 one gcp was used on average every ~3900ha. By increasing the number of gcps for job 676 from 19 to 30 resulted in one gcp for every 3900ha instead of one every ~6100ha.

3.2.2 Aerial triangulation with bundle block adjustment

Aerial triangulation in the form of bundle block adjustment using a rigorous camera model was conducted to compute the exterior orientation of individual images in all four historical aerial image blocks. The bundled triangulation approach determines the exterior orientation using the collinearity condition as the basis for formulating the relationship between the images contained within each block, the camera model, and the ground (Kaichang et al. 2003; Schenk 2004; Nagarajan & Schenk 2016). Collinearity is built on the condition that the image's perspective centre, ground and corresponding image point are in a straight line and can be expressed by the following equations:

$$x - x_o = -f \frac{a_{11}(X - X_s) + a_{12}(Y - Y_s) + a_{13}(Z - Z_s)}{a_{31}(X - X_s) + a_{32}(Y - Y_s) + a_{33}(Z - Z_s)} \quad (1)$$

$$y - y_o = -f \frac{a_{21}(X - X_s) + a_{22}(Y - Y_s) + a_{23}(Z - Z_s)}{a_{31}(X - X_s) + a_{32}(Y - Y_s) + a_{33}(Z - Z_s)} \quad (2)$$

where X, Y and Z are the ground coordinates, x and y are the image coordinates, X_s, Y_s and Z_s are the coordinates of the exposure centre in the ground coordinate system, f is the calibrated focal length of the camera, x_o and y_o are the image coordinates of the principle point, and a_{ij} are the elements of the rotation matrix with the three angles (ω, φ, k) (Omega, Phi and Kappa – yaw, pitch and roll). f, x_o and y_o represent the interior orientation parameters which were measured using the camera calibration report, while X_s, Y_s, Z_s, $\omega, \varphi, \text{and } k$ are the exterior orientation parameters (Kaichang et al. 2003; Schenk 2004; Nagarajan & Schenk 2016).

In the context of bundled adjustment computation, the collinearity equations are generated and solved in the form of a functional model, dependent on the number of overlaps between individual images (created by tie points) as well as overlaps with the ground control network. A number of observation equations are formulated and the collinearity condition is solved in a bundled solution using a least squares adjustment to estimate and adjust values associated with the exterior orientation while minimising and distributing error through the network of observations (Raj 2006; Nagarajan & Schenk 2016). Essentially this comprises reconstructing the 3D position and 3D rotation of each aerial image's perspective centre based on the camera information, interior orientation parameters, approximate relative image position and ground control points. After aerial triangulation with a bundled adjustment is performed, the differences between the initial measurements and the new estimates are the residuals. These provide a preliminary indication of the accuracy of the solution that has been formulated and applied (Kaichang et al. 2003; Raj 2006).

For this study, warped images necessitated manual interior orientation to align image and camera coordinates using the fiducial marks present on the images. Automatic tie point extraction was achieved using a combination of feature-based and least-squares matching procedures but was only possible after the relative positions of the images in the block were refined by placing three manual tie points in *Von Gruber* positions for each image

(Trimble/Inpho, 2014a). Several post-processing models were run on the results of the functional model used to triangulate each photo job. This was undertaken to gauge the most effective post-processing procedure in terms of highest accuracy achieved within the aerial triangulation and final orthorectification process. Post-processing parameters adjusted were associated with blunder removal, self-calibration, changes in the standard deviation settings of ground control points and the level of automated versus manual points included in the model. Self-calibration creates self-calibrated camera parameters based on the results from the rigorous camera model and additional control points evenly distributed as grid points across an image. It essentially accounts for the systematic errors associated with camera interior geometry.

Initially it was challenging to determine a relationship between any of the post-processing model runs in terms of aerial triangulation accuracy and the final accuracy in ground control positioning in the orthoimage generated. This is because error shifted between different sets of ground control points unevenly between model runs i.e. the error at a particular ground control point could be the lowest in one model run and then the highest in another and this differed substantially across photos in a specific photo job. This was particularly evident when varying ground control standard deviations. For example, certain areas across a photo job and within individual photos would become more accurate while other areas in the job would experience greatly increased error. Furthermore, the aerial triangulation accuracy results were not entirely useful for understanding final planimetric accuracy and adding manual points seemed to decrease the aerial triangulation accuracy results. An iterative process was used to narrow down the different types of post-processing iterations into eight different models, which adequately displayed effects on aerial triangulation results and were assumed, through a basic visual inspection, to influence final planimetric accuracy (Table 2).

Table 2 A list of post-processing models that were selected and run to evaluate the success of aerial triangulation accuracy and final orthorectification accuracy.

Model name	**Self-calibration**	**Blunder removal**	**Eliminate manual points**	**Standard deviations ground control (planimetric, height [m])**	**Automatic and manual tie points**	**Standard deviations for automatic and manual tie points [μ (pixels)]**
std0.1	off	off	no	0.1, 0.1	yes	4 (0.2), 10 (0.5)
b0.1	off	radicle	yes	0.1, 0.1	yes	4 (0.2), 10 (0.5)
c0.1	44	off	no	0.1, 0.1	yes	4 (0.2), 10 (0.5)
cb0.1	44	radicle	yes	0.1, 0.1	yes	4 (0.2), 10 (0.5)
no-auto0.1	off	off	no	0.1, 0.1	manual only	4 (0.2), 10 (0.5)
std2-5	off	off	no	2, 5	yes	4 (0.2), 10 (0.5)
c2-5	44	off	no	2, 5	yes	4 (0.2), 10 (0.5)
cb2-5	44	radicle	yes	2, 5	yes	4 (0.2), 10 (0.5)

The procedure started with a standard model run (std0.1), which included no blunder removal, no self-calibration and with fixed standard deviations set for planimetric and height ground control points (namely, 0.1 and 0.1). This standard model run was then post processed using the seven different options described in Table 2. This resulted in eight different post-processing options being run on four different photo jobs, which differed in terms of the number of images, area covered, and ground control points used. Standard deviations for automatic and manual image measurements were not changed between the different post-processing model runs and were set at the recommended default values.

For each post-processing run, the accuracy results of the aerial triangulation with bundle block adjustment generated by the MATCH-AT (Trimble/Inpho 2014a) were captured and imported into RStudio for analysing potential correlations with orthorectification accuracy results. The residuals specifically indicate the degree to which an observation (input) fits with the functional model and includes error estimates for automatic and manual tie points, ground control points and an overall accuracy estimate (Table 3).

Table 3 Accuracy parameters generated in Inpho photogrammetric software for the aerial triangulation process with bundle block adjustment.

Abbreviation (used in paper)	General name	Description	Ideal value
Sig0	Sigma0 [microns]	An overall accuracy result for the fit of the block considering manual and/or automatic points and ground control points (XYZ)	1/3 of a pixel (therefore 7.05 µ for this study)
manual.x/y	RMS manual points in photo	Residuals in manual tie points for x and y	Dependent on the standard deviations set. The default standard deviations were used for this study
auto.x/y	RMS automatic points in photo	Residuals in automatic tie points for x and y	
at.XY/Z	RMS control points	The total root mean square error of the residuals in meters of X, Y and Z ground control points	Dependent on standard deviations set. Ideally this should be less than or roughly equal to the standard deviations set for the project
at.max.XY	Maximum RMS control points XY	The maximum root mean square error of the residuals in meters of X and Y ground control points	

3.2.3 Orthorectification

After each post-processing run, the exterior orientation for each photo job was imported into OrthoMaster. Orthographic corrections were performed using the Ortho Rectification tool and a 25m Digital Elevation Model (absolute accuracy 2.5 – 5m) (Trimble/Inpho 2014a).

3.2.4 Accuracy assessment

For assessing final orthoimage planimetric accuracy, orthoimages generated in OrthoMaster and ground control points captured in the current orthoimage aerial triangulation project files were imported into ArcGIS. A manual procedure was then used to plot the deviation of each control point in the historical orthoimages from the ground control point in the current orthoimages (reference images) i.e. the Pythagorean hypotenuse of the right angle triangle made up of the residuals in the x- and y-axes (Rocchini et al. 2012). At times, more than one deviation had to be measured for an individual control point. This resulted because of the overlaps between images and image strips. For example, for photo job 266 there were 40 ground control points with 117 photos covering at least one of these. Thereby resulting in 117 deviations being measured between the ground control points on the current orthoimages and their positions on the historical orthoimages. This "deviation" is the root mean square error (RMSE) and can be expressed as follows:

$$RMSE_i = \sqrt{{x_i}^2 + {y_i}^2} \qquad (3)$$

where i is each photo that overlaps with a ground control point and x_i and y_i are the residuals in the x and y axes. The total root mean square error for each post-processing run was derived using the following equation.

$$Total\ RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n} RMSE_i} \qquad (4)$$

where $RMSE_i$ is the error associated with each ith photo that overlaps a ground control point (Rocchini et al. 2012) (Table 4).

The root mean square error distances and the direction (of the error) were captured in table format in ArcGIS and imported into R Studio. RMSE and associated directions of error were plotted using ggplot2 in Rstudio (Wickham 2009; R Core Team 2016). Once results had been generated for all post-processing models, planimetric accuracy was then assessed at checkpoints for all photo jobs but only for the top two performing models and the standard model run (std0.1)

Spearman rank correlation was used to investigate correlations between accuracy results from the aerial triangulation process (Table 3) and final planimetric accuracy (Table 4) of orthoimages generated. Specifically correlations were investigated between final aerial triangulation with bundle block adjustment accuracy results for each post-processing model run (i.e. Sig0, manual.x/y, auto.x/y, at.XY, at.max.XY, and at.Z) and final orthoimage planimetric accuracy (i.e. or.XY and or.max.XY) (Wickham 2009; Wei 2013; Harrell & Dupont 2015; R Core Team 2016). Post-processing models were split into two groups when investigating correlations between RMS control points (at.XY/Z) and final orthoimage planimetric errors. This is because the standard deviations set for planimetric and height control points differed between post-processing models and this influences the RMS results for control points in the aerial triangulation process.

Table 4 Accuracy assessment variables derived from manually measured deviations between historical orthoimages generated and reference orthoimages in ArcGIS.

Abbreviation (used in paper)	General name	Description	Ideal value
or.XY	RMSE control points XY	The total root mean square error of residuals between ground control points in final historical orthoimages and current reference orthoimages.	As low as possible but dependent on study application. RMSE of ~ 10m have been achieved for other studies working in rugged terrain with <4 for more gentle terrain (Wang & Ellis 2005b; Rocchini et al. 2006, 2012)
Or.max.XY	Maximum RMSE control points XY	The maximum root mean square error found between ground control points in final historical orthoimages and current reference orthoimages	As low as possible but dependent on study application. ~15m has been achieved for other studies working in rugged terrain (Rocchini et al. 2012) .

4. Results and Discussion

4.1 Planimetric accuracy of orthoimages

The self-calibrated post-processing model, c0.1, consistently showed the highest final planimetric accuracy at ground control points for all four historical aerial photo jobs (Figure 7). This model included manual and automated tie points and ground control standard deviations of 0.1m and can be viewed as achieving reasonably high accuracy especially given the limitations of the study area (complex geomorphology) and images (warpage across image surfaces) (Rocchini et al. 2012). The model also managed to retain the accuracy achieved at ground control for additional checkpoints measured (i.e. for points not included in the aerial triangulation process) (Figure 8).

The non-automated model, which only included manual tie points (i.e. noauto0.1), showed the next highest accuracy at ground control for three of the photo jobs (226, 225 and 699). For the fourth photo job (676), the non-automated model was on par with the self-calibrated model, cb2-5, which was flexible around the ground control at a standard deviation of 2m and 5m respectively for planimetric and height coordinates (Figure 7). In contrast to the self-calibrated model, the non-automated model showed inconsistent planimetric accuracy across orthoimages with three out the four photo jobs showing decreased planimetric accuracy at additional checkpoints in terms of total and maximum root mean square errors. In certain cases, this included an increase of more than 30 meters of planimetric error (Figure 8).

Many of the other post-processing models resulted in extremely large and variable final planimetric error similar to what one may expect from a polynomial rectification approach in mountainous regions (Figure 7). The standard model, std0.1, which included no blunder detection, and therefore all manual tie point and ground control point measurements retained in the adjustment, consistently achieved higher final orthoimage planimetric accuracy in comparison to blunder detection, b0.1, post-processing model. This is an indication that the process was more effective when all manual points and ground control points were included as opposed to certain of these points deleted, in an attempt to create a "best-fit" model. Post-processing models that allowed the aerial triangulation process more room to adjust around ground control points (i.e. setting standard deviation of five and two meters) were ineffective at reducing the final root mean square errors in orthoimages. This is despite being a far more realistic standard deviation estimate. It is rather unrealistic to assume the standard deviation for ground control digitised from an orthoimage as being within 10cm. Constraining the standard deviations, however, clearly resulted in higher planimetric accuracy.

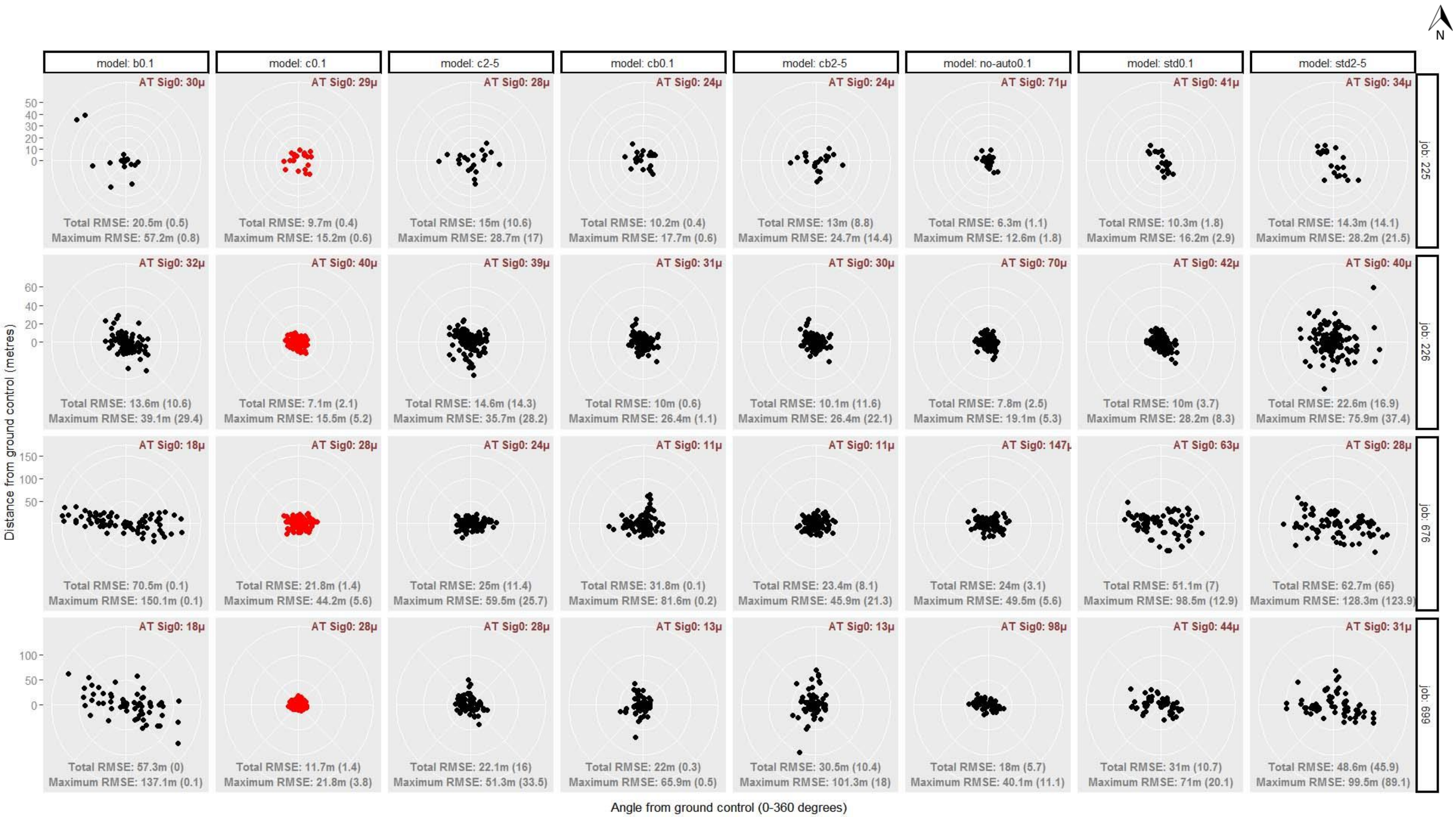


Figure 7 Final planimetric accuracy of orthoimages generated i.e. root mean square errors (RMSE) and direction of error for all ground control points in final orthoimages generated for the four photos jobs. Final planimetic accuracy is summarised by the total and maximum root mean sqaure error (RMSE) indicated at the bottom of each polar plot. Aerial triangulation accuracy measurements are included for comparison purposes including total and maximum XY root mean square error results shown in brackets and the overall triangulation accuracy (AT Sig0) .

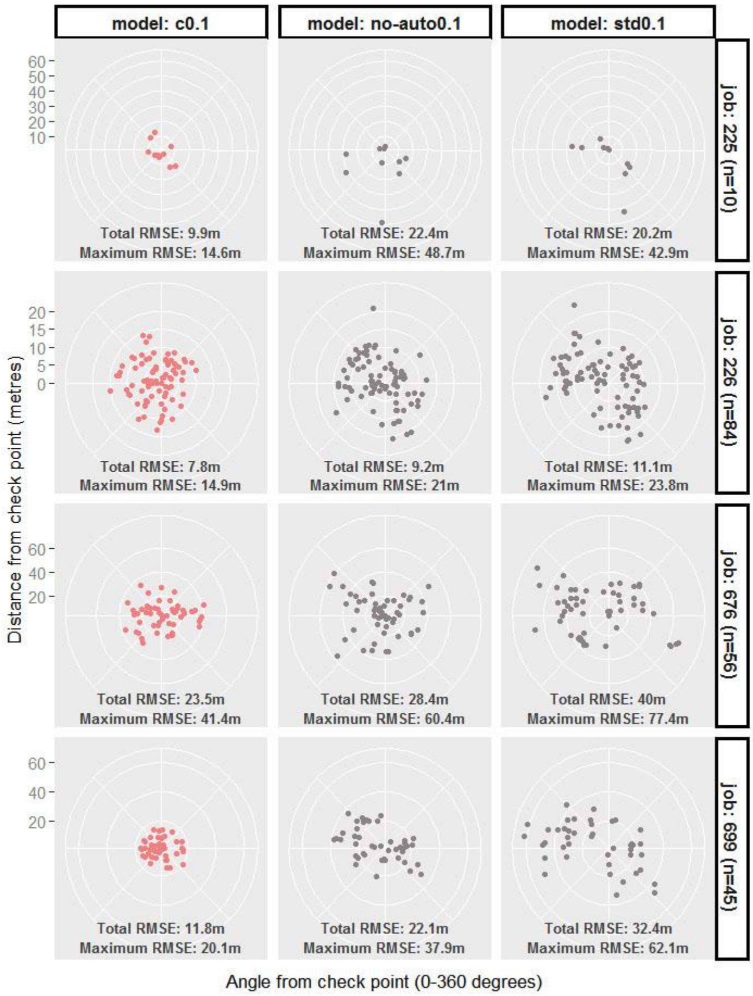


Figure 8 Final planimetric accuracy at check points for orthoimages for all four photo jobs for c0.1, no-auto0.1 and std0.1 post-processing models.

4.2 Aerial triangulation accuracy and correlations with final planimetric accuracy

The overall aerial triangulation accuracy for most post-processing models was much lower than the recommended one third of a pixel. Average Sigma0 was 38 µ (1.8 pixels) and ranged from 11µ (0.5 pixels) to 147µ (7 pixels) (see *AT Sig0* in red in Figure 7). However, estimating final orthoimage planimetric accuracy using the aerial triangulation results was not an intuitive process (Figure 7 and Figure 9). For example, the overall accuracy of the aerial triangulation process as measured by the Sigma0 was negatively correlated with final planimetric accuracy at ground control i.e. with the total and maximum root mean square errors measured in the final orthoimages (or.XY and or.max.XY) (Figure 9 and Figure 10 A). When Sigma0 was high, final planimetric accuracy was low. Ideally, you would expect to see a positive correlation, as the lower the Sigma0 the more accurate the model run for the aerial triangulation and therefore a lower root mean square error is assumed for ground control in final orthoimages generated. What the results show is the opposite effect. A similar relationship was found for residuals in automatic points x (auto.x) (Figure 9 and Figure 10 B). No correlations were found between other aerial triangulation accuracy results related to residuals in automatic and manual points and final planimetric accuracy in historical orthoimages (Figure 9).

When considering aerial triangulation accuracy results for ground control there were, again, no correlations present for post-processing models, which were set at a standard deviation of 0.1m (Figure 11 A). For more flexible models (standard deviation at 2m and 5m), however a more positive relationship seemed to develop showing positive correlations between maximum planimetric accuracy in orthoimages (or.max.XY) and maximum aerial triangulation root mean square errors and residuals for height ground control (Figure 11 B). However, these models also showed the lowest final planimetric accuracy and therefore this is not an entirely useful measurement.

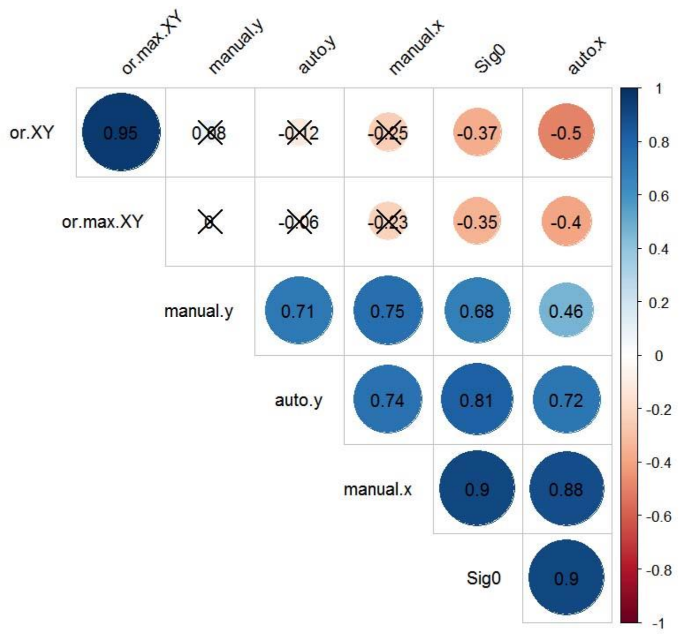


Figure 9 Spearman rank correlation coefficients between final orthoimage planimetric accuracy (or.XY and or.max.XY) and aerial triangulation accuracy measurements for overall accuracy (Sig0), manual tie points (manual.x and manual.y), and automatic tie points (auto.x, auto.y). For all pairwise comparisons n=32 (i.e. 8 models * four photo jobs) except where the pairwise comparison included models run with no automatic points whereby n=28 (p values >0.05 are indicated by a cross through the coefficient values).

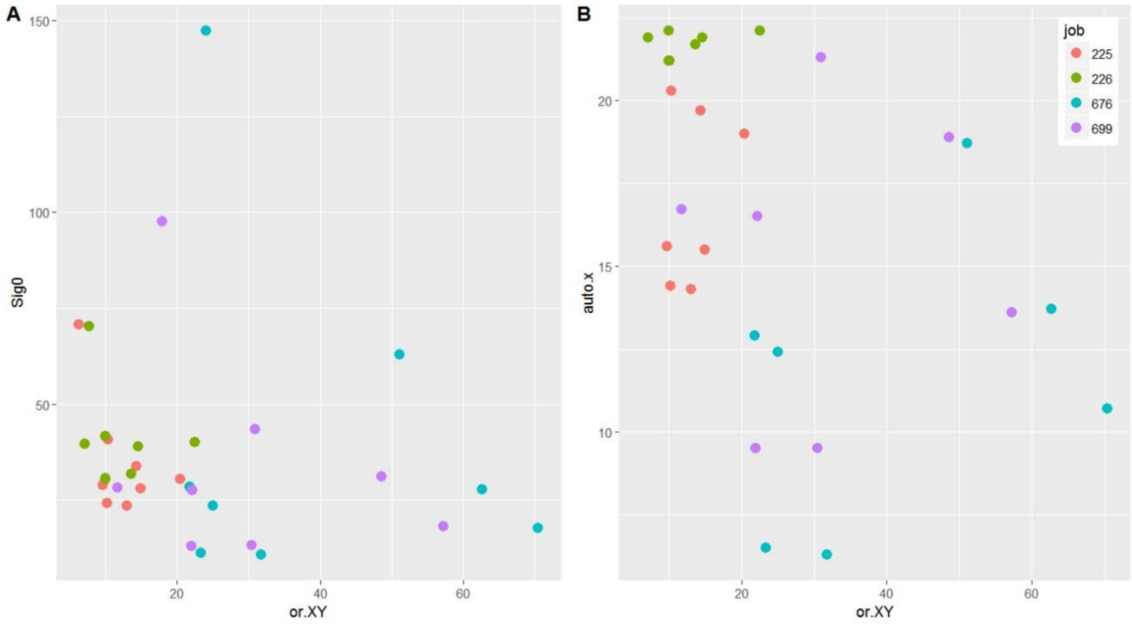


Figure 10 Scatter graphs showing weak negative correlations between planimetric accuracy of final orthoimages (or.XY) and Sigma0 [μ] in graph A; and aerial triangulation root mean square errors for automatic points x (auto.x) in graph B.

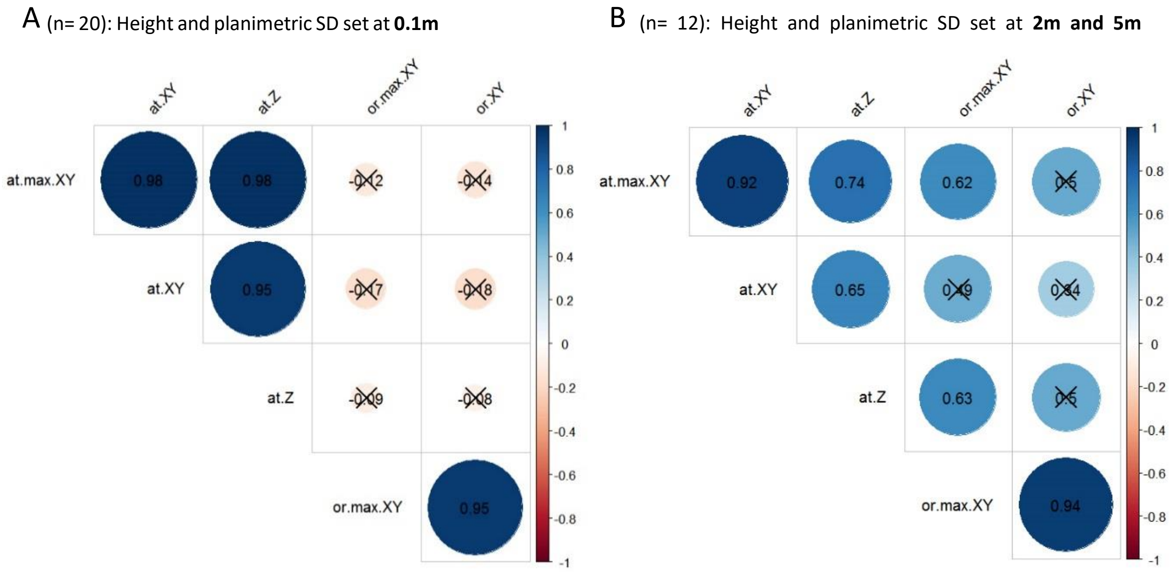


Figure 11 Spearman rank correlation coefficients between final orthoimage planimetric accuracy (or.XY and or.max.XY) and aerial triangulation root mean sqaure errors at ground control (at.XY and at.Z) for post-processing models grouped by aerial triangulation ground control standard deviation (SD) settings. 0.1m SD for both planimetry and height shown in corrplot A; and 2m and 5m SD for planimetry and height respectively shown in corrplot B (p values >0.05 are indicated by a cross through the coefficient values).

Poor overall aerial triangulation results as well as mostly poor final planimetric accuracy and limited and negative correlations between aerial triangulation results and final planimetric accuracy could be linked to a combination of aspects. It is, however, likely that the degraded image quality played a role as this would have resulted in large and highly variable accuracy results for the interior orientation of all images i.e. the first step in the aerial triangulation process. Interior orientation computes the transformation parameters from the pixel coordinate system (row, column) into the image coordinate system (x, y). It establishes a basis to which all following processes refer. Only a high-quality interior orientation can produce subsequent high accuracy results. Sigma0 for interior orientation ranged between 4 and 15 pixels (Figure 12, Figure 13 and Table 5). This is considerably poor in comparison to the recommended one third of a pixel.

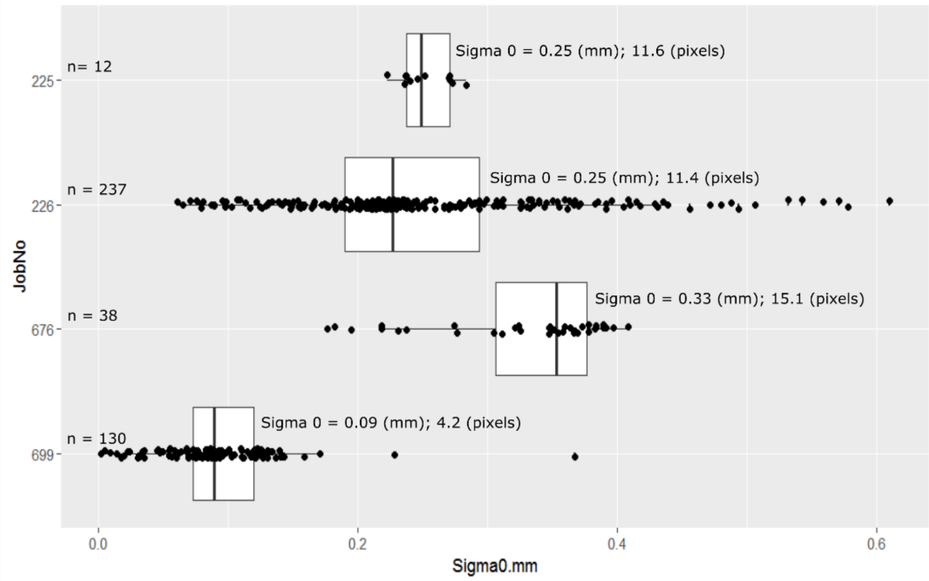


Figure 12 Boxplot of interior orientation results in mm for the four different aerial photos jobs. Jitter points indicate individual images per photo job. The number of images (n) and the mean Sigma0 (in mm and pixels) is shown for each photo job.

Poor interior orientation because of warpage and degradation across the entire image surface could have reduced the overall accuracy of the aerial triangulation process for all calibrated as well as non-calibrated models. It is also likely that the poor interior orientation process prevented a fully automated tie point matching process and the need for manually placing points in the *Von Gruber* positions for all images in all photo jobs. Poor interior orientation likely resulted in large residuals of the corresponding image and object coordinates. Therefore, although manually and automatically measured points were visually correctly measured, when using blunder detection, they were eliminated because of poor transformation parameters of the interior orientation (Table 6). Furthermore, for post-processing models in which blunder detection was not used the accuracy calculated for image and object coordinates was low, and this affected the overall Sigma0 and resulted in misalignment between aerial triangulation results and final planimetric accuracy in orthoimages.

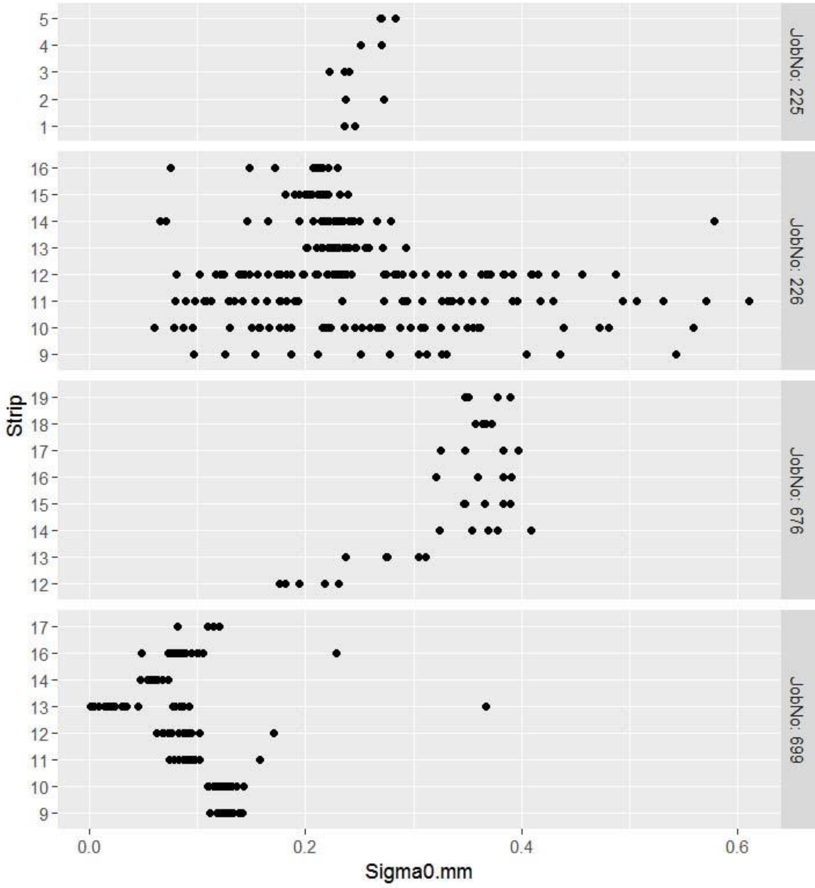


Figure 13 The range of interior orientation results (Sigma 0) displayed for each strip per photo job. Jitter represents individual images. Showing the spread of error across images and strips.

Blunder detection removes observations based on standard deviations specified. The smaller the standard deviation specified the higher the weight and the more accurate observations must be, not to be eliminated. The higher the standard deviation values, larger observation error is accepted, and the observation will not be eliminated. Therefore, when blunder detection was used in combination with a standard deviation of 0.1m the process removed manually placed points (including ground control), which were correctly placed. This was then determined by the aerial triangulation model as an increase in accuracy whereby “incorrect” tie points or control points (which in fact were correct) had been removed. As a result, post-processing models, including blunder detection and fixed to the ground control showed some of the highest accuracy for the aerial triangulation process i.e. lower Sigma0 and ground control root mean square errors (e.g. b0.1 and cb0.1). This however did not translate into similar levels of accuracy in the final orthoimages generated as achieved when using the self-calibrated model, c0.1, which showed lower aerial triangulation accuracy but higher final orthoimage planimetric accuracy at ground control (see b0.1 and c0.1 in Figure 7). One way of avoiding manual and automatic measurements from being eliminated is by giving them a larger standard deviation, thus allowing bigger residuals. However, post-processing models with planimetric and height standard deviations of five and two meters respectively did not achieve acceptable levels of final planimetric accuracy (Figure 7).

Table 5 Images at the minimum and maximum interior orientation error extremes for photo job 226. Deformations are easier to pick up along the edges and especially for image 226_11_4525, which receieved the lowest accuracy result for interior orientation out of all the images across the four photo jobs.

Job No 226

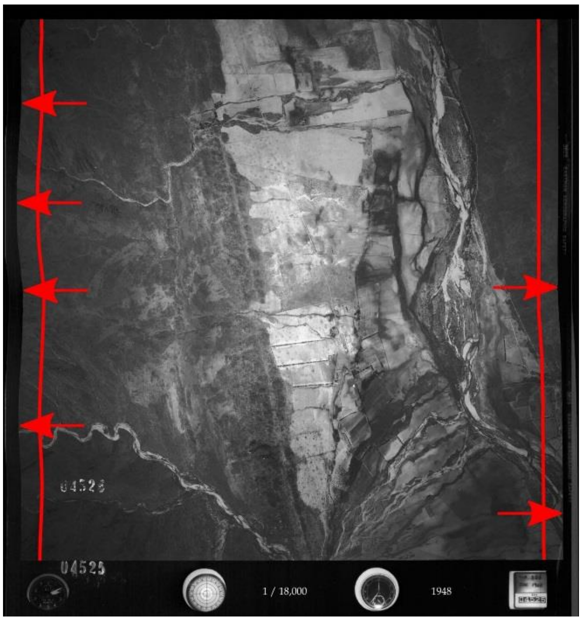
Min Sigma0: 0.06 mm	Max Sigma0: 0.61 mm
	
226_10_4487	226_11_4525

Table 6 Resulting number of ground control points, automatic tie points and manual points for the different post-processing runs. Models which included blunder detection decreased the number of points used for the aerial triangulation process percieving them to be incorrect.

Photo job	225	226	676	699	225	226	676	699	225	226	676	699
Post-processing model	No. of ground control				No. of automatic tie points				No. of manual tie points			
b0.1	6	13	3	4	887	26769	6106	16941	95	4063	397	1654
cb0.1	8	16	9	6	884	26773	5713	16558	97	4063	395	1595
cb2-5	9	40	30	21	884	26766	5755	16599	100	4078	453	1632
no-auto0.1	9	40	30	23	0	0	0	0	107	4521	521	1913
c0.1	9	40	30	23	891	26778	6419	17968	107	4521	521	1913
c2-5	9	40	30	23	891	26778	6419	17968	107	4521	521	1913
std0.1	9	40	30	23	891	26778	6419	17968	107	4521	521	1913
std2-5	9	40	30	23	891	26778	6419	17968	107	4521	521	1913

4.3 Discussion

The self-calibration process in combination with constrained standard deviations of 0.1m consistently resulted in improved final planimetric accuracy across the four different photo jobs tested which included different ground control points, area coverage and number of images. Although the process could not be fully automated, a substantially reduced number of ground control points was used in all photo jobs in comparison to the recommended 4-20 per image or image pair. For example, a photo job of 200 images would require at least 400-800 ground control points being identified on both the historical image but also on the current reference image as well as at least 600 tie points being cross-referenced across overlapping photo pairs (depending on the georeferencing process being used). This study used 40 ground control points for a photo job of 237 images (covering ~150 000 ha) and placed three manual tie points per image. Although, manual tie points between images were required these are much quicker to find and measure in comparison to ground control points.

Errors from warpage are cumulative during the aerial triangulation and orthorectification process. Therefore, inherent deformations can negatively influence the aerial triangulation process especially in terms of the level of automation that can be achieved. This study has shown however that it is not completely restrictive with certain post-processing models (and especially the c0.1 model) still being able to achieve relatively acceptable accuracy. Based on the results of this study, it seems necessary to include manual points in *Von Gruber* positions depending on interior orientation results, but also to use automated tie point extraction for extra connections between photos and photo strips. Although the aerial triangulation results should be reviewed it should be noted that these are highly influenced by the standard deviations set as well as other post-processing parameters linked to blunder detection and calibration. Therefore, high aerial triangulation accuracy does not always translate into high planimetric accuracy in final orthoimages generated when considering historical aerial imagery.

Error in the orthorectification process that can affect the planimetric position of objects is also dependent on the accuracy of the DEM as well as the measurement of control points. In this study, this source of error was kept constant across all post-processing model runs. Furthermore, control points were measured from current orthoimage digital aerial triangulation files, which were based on a photogrammetrically compiled DEM, which had an absolute accuracy of 2.5 to 5m. This photogrammetrically compiled DEM was also used to generate the historical orthoimages. Therefore, any error in the DEM should be reflected both in the current reference orthoimage and therefore in the historical orthoimage generated. These should therefore be aligned.

Self-calibration is recommended as an option to assist in reducing interior orientation problems inherent when working with historical aerial images. In the past self-calibration has been used to compensate for effects like film stretching during the development process but can also be used to compensate for poor scanner calibration. However, it is usually recommended to correct image measurements for systematic effects (Kaichang et al. 2003). The warpage that is evident in the historical aerial images used in this study is presumed to be randomly spread across the image surface and therefore random between image models within a strip and between flight strips. Despite the assumed random nature of the error, self-calibration has proved effective for increasing final planimetric accuracy to acceptable levels at both ground control and furthermore at additional check points indicating that there may have been some level of systematic error in the aerial images.

Orthoimages generated using the c0.1 post-processing model are acceptable for area based changed detection as well as relative proportions of change in terms of the total area under study. This is because the geometry of features still seems intact. The planimetric placement of features in terms of the current imagery is however inaccurate by 7.1 to 21.8m (based on the total RMSE) depending on the photo job being used. This prevents overlaying the historical orthoimages onto the current reference orthoimages for direct pixel-to-pixel or even area-to-area comparisons. A major advantage of the digital aerial triangulation process with bundle block adjustment is that individuals interested in the study area can pick up on where others left off i.e. increase the accuracy of the orthoimages by working on the existing aerial triangulation project file which already includes the manual tie point placements. The influence of the accuracy and number of points in the control network is a theme for further research. Furthermore, it is possible that the addition of more manual tie points for post-processing models that do not include blunder detection could reduce final planimetric errors in orthoimages. However, this would likely reduce the aerial triangulation accuracy results, an artefact of the post-processing model assuming manually placed points are incorrectly placed because of poor interior orientation, but also increase the time required for each independent photo job.

4.4 Comparison to international and local studies

Despite the overall poor interior orientation and aerial triangulation accuracy results, the total and maximum root mean square errors achieved for ground control and checkpoints for the constrained self-calibrated model, c0.1, are on par with orthorectification results achieved in other geomorphologically complex areas with higher quality original analogue photos and better scanning procedures. Rocchini et al. (2012) achieved total and maximum root mean square errors of 9.1 and 14.22 m respectively for historical aerial photos covering Monte Baldo, Province of Trento, in the Italian Alps environment using approximately 16 ground control points per image. Errors, however, are still larger in comparison to work conducted in less complex landscapes and which included far more ground control points (Wang & Ellis 2005b). For example root mean square errors were kept under 4m for work done on the slopes of Mt Amiata where elevation ranges from 664-1016 m and whereby the study used approximately 30 ground control points for an area covering 440ha (Rocchini et al. 2006). RMSE were also kept under 4m for work conducted in rural sites of 10 000ha in China ranging from 5-570m in elevation and whereby 30-60 ground control points were used (Wang & Ellis 2005b).

Similarly, to international studies, change detection studies in South Africa have focused on using 4-20 ground control points per image to conduct georeferencing and orthorectification. As a result, many studies use a small number of images and focus on a relatively small area of interest. In addition, studies in South Africa using historical aerial images have mainly relied on polynomial functions. The use of polynomial functions is assumed to be driven by the fact that many studies have been in areas of low relief. Generally local studies tend to not report on georeferencing error with most (if not all) studies dismissing the need for checkpoints and therefore rendering the total root mean square errors (when reported) an inadequate reflection of the positional error across the images. In studies that have reported errors, these have been relatively large, despite the flat terrain, ranging from ±15 – 30 meters (total RMSE). This is substantially greater than the RMSE achieved in this study for the c0.1 post-processing model which was used for a photo job of over 200 photos covering extremely complex terrain (Hudak & Wessman 1998; Higgins et al. 2001; de Neergaard et al. 2005; Garden & Garland 2005; Keay-Bright & Boardman 2006; Giannecchini et al. 2007; Wigley et al. 2009, 2010; Corrigan et al. 2010; Palmer et al. 2010; Grenfell et al. 2010; Puttick et al. 2011, 2014; Buitenwerf et al. 2012; Gordijn et al. 2012; Halpern & Meadows 2013).

5. Conclusion

Globally, advanced methodologies have been used for preparing historical images for studying changes in landscapes. This includes automating parts of the georeferencing process and creating mosaics using feature extraction and matching algorithms. However, for single-image orthorectification (and hence georeferencing) at least four and up to 20 ground control points as well as (depending on the methodology) six tie points are manually measured. This is a considerable amount of manual effort. As a result, many orthorectification studies either consider a low number of photos, a comparatively small area and an extensive ground control network (Ellis et al. 2006).

This study has shown that automatic aerial triangulation can be used for orthorectifying historical aerial photos with a reduced number of ground control for large blocks of aerial imagery. Self-calibration in conjunction with constraining standard deviations around the ground control proved an important component of the process for achieving acceptable levels of planimetric accuracy in final orthoimages comparable to other studies working in geomorphologically complex areas as well as an improvement to georeferencing studies in South Africa. This has been tested for an extremely mountainous region in South Africa with four photo jobs (ranging from 12 to 237 images) including images with inherent deformations linked to film warpage and shrinkage and which cover a large area (ranging from >33 000 ha to approximately 150 000ha). High aerial triangulation accuracy did not always translate into high planimetric accuracy in final orthoimages generated. This was linked to standard deviations set for image and object coordinates, and the removal of points and hidden errors associated with blunder detection and self-calibration post-processing models. Future work should consider the effects of increasing the accuracy and changing the number and placement of ground control and manual tie points.

The national mapping department in South Africa has an extensive collection of historical analogue aerial photos with a large coverage across the country dating from the early 1900s. This is not unique to South Africa with many other countries (including least developed and developing) also having extensive analogue photos in storage (Tekle & Hedlund 2000; Palandro et al. 2003; Schiefer & Gilbert 2007). NGI also has a collection of current (2008-2014) colour (RGB) and colour infrared (CIR) digital ortho aerial photos covering the whole country. These digital orthoimages provide a resource for referencing historical imagery, especially for extracting ground control points. Ideally, imagery could be provided on an open source platform ready for viewing by the public and for review for potential change detection studies by applied scientists and other researchers. GeoMemories is an example of an initiative working towards this goal in the Italian landscape (Abrate et al. 2013). Libraries of Brock University, Santa Barbara and Stanford Universities have also been conducting similar work towards creating portals for national aerial photo archival data collection and distribution (Ma & Buchwald 2012).

Appendix 5.2 Hydrological response units, irrigated areas, impervious areas and dams (Table 1 and 2), parameters for land use/cover specific variables for hydrological response units and irrigated areas (Table 3), model configuration (Figure 1) and annual flow duration curves (Figure 2 and 3)

Table 1 Total area (in hectares) for hydrological response units, irrigated areas, impervious areas to rainfall and dams outside and inside the protected area for four land use/cover scenarios and a natural scenario. For modelling purposes, areas were calculated for each Quinary sub-catchment portions shown in Figure 1 below. Areas < 1 ha within a Quinary sub-catchment portion were not included in the model with these areas being incorporated as natural fynbos i.e. the fynbos hydrological response unit.

		Outside protected area					Inside protected area				
Hydrological modelling units		Natural	1949	1972	LB2014	HB2014	Natural	1949	1972	LB2014	HB2014
HRU	**Hydrological response units**										
1	Fynbos	**15 878**	**10 304**	**10 922**	**14 411**	**2575**	**15 512**	**10 442**	**11 952**	**15 246**	**1272**
2	Grazing	0	**5279**	**4179**	**716**		**0**	**5017**	**3373**	0	
3	Low burn fynbos	0	0	0	**3**	0	0	0	0	**247**	0
4	High burn fynbos	0	0	0	0	**11839**	0	0	0	0	**14 221**
5	Dense aliens	0	0	**127**	**8**		0	0	0	0	
6	Scattered aliens	0	**10**	**71**	**23**		**0**	**1**	**9**	**1**	
7	Dryland farming (rooibos and other)	0	**72**	**19**	**55**	**55**	**0**	**8**	**4**	0	
IA	**Irrigated areas**										
1	Blue and raspberries	0	0	0	**120**	**120**	0	0	0	0	0
2	Peaches and/or apples	0	**120**	**381**	**16**	**16**	0	**10**	**104**	0	0
3	Proteas	0	0	**5**	**269**	**269**	0	0	0	0	0
4	Olives	0	0	0	**16**	**16**	0	0	0	0	0
5	Buchu or other oils	0	0	0	**22**	**22**	0	0	0	0	0
6	Tobacco	0	**25**	**28**	0	0	0	0	0	0	0
7	Artificial pastures	0	0	**71**	0	0	0	0	**12**	0	0
8	Nuts	0	**21**	0	**24**	**24**	0	**3**	**1**	0	0
9	Annual crops (sweet potatoes and beans)	0	**11**	0	0	0	0	**21**	**33**	0	0
10	Citrus	0	0	**11**	**57**	**57**	0	0	**1**	0	0
O	**Other**										
1	Disjunct impervious areas	0	**35**	**59**	**96**		**0**	**9**	**23**	**17**	
2	Dams	0	**0.3**	**5.3**	**42**		**0**	**0.07**	**0.25**	**0.06**	

Table 2 Change in number and total surface area of dams determined using historical and current orthoimages and the calculated total capacity using all the shapes equation (5.2.3.3.5).

Quinary portions		1949	1972	2014	1949	1972	2014	1949	1972	2014
		Total no. dams			**Total surface area (ha)**			**Total capacity (m³)**		
Outside protected area										
E10C2	sub3	2	13	26	0.025	4	32	82	40 301	700 609
E10C3	sub2	0	1	3	0	0.03	5	0	151	87 940
G10G2	sub1	0	0	1	0	0	0.096	0	0	476
G10H1	sub2	3	5	7	0.25	1	4.6	1543	9930	54 757
Inside protected area										
G10G1	sub1	0	1	0	0	0.076	0	0	427	0
G10G2	sub1	1	2	1	0.066	0.14	0.06	356	899	295
G10H1	Sub1	0	1	0	0	0.02	0	0	114	0

Table 3 Monthly values of water use coefficients (CAY), canopy interception per rain day (VEGINT), root mass distribution in the topsoil (ROOTA), coefficient of initial abstractions (COAIM), index of suppression of soil-water evaporation by a litter/mulch layer (PCSUCO), and the percentage of roots colonising the subsoils horizon (COLON) for hydrological response units and irrigated occurring outside and inside the protected area.

Model component		Para-meters	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Source and description, including ACRU code where relevant
			Monthly values												
Hydrological Response Units (HRUs)															
1	Fynbos	CAY	0.45	0.45	0.5	0.6	0.6	0.6	0.6	0.6	0.6	0.55	0.5	0.45	Macchia (Acocks # 69) [ACRU CODE: 2020102]
		VEGINT	1	1	1.1	1.2	1.2	1.2	1.2	1.2	1.1	1.1	1	1	
		ROOTA	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	
		COIAM	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
		PCSUCO	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	
		COLON	100	100	100	100	100	100	100	100	100	100	100	100	
2	Grazing	CAY	0.3	0.3	0.35	0.45	0.45	0.45	0.45	0.45	0.45	0.4	0.35	0.3	For HRUs 2-4, CAY, ROOTA, COIAM, PCSUCO and COLON are based on those derived for moderately and severely degraded vegetation types from False Macchia [ACRU CODE: 2020103] e.g. Kouga, Elandsberg, Langkloof, Groendal, Gamtoos and Tsitsikama variations of fynbos. All fynbos variations have the same coefficients as False Macchia, which share the same coefficients as Macchia. All degraded forms share the same coefficients for these two levels of degradation. Macchia and False Macchia only differ in terms of interception losses whereby Macchia has higher interception losses. Therefore, VEGINT is reduced by 30, 20 and 50% for grazing, low intensity burn area and high intensity burn area respectively. COLON is reduced for grazing and high burn due to removal of above and below ground plant material as well as poor root development in the case of grazing. Changes made are based on recommendations in Smithers & Schulze (1995) and information presented in Table 5.3 in Section 5.
		VEGINT	0.7	0.7	0.77	0.84	0.84	0.84	0.84	0.84	0.77	0.77	0.7	0.7	
		ROOTA	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	
		COIAM	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	
		PCSUCO	10	10	10	10	10	10	10	10	10	10	10	10	
		COLON	40	40	40	40	40	40	40	40	40	40	40	40	
3	Low intensity burn area	CAY	0.35	0.35	0.40	0.50	0.50	0.50	0.50	0.50	0.50	0.45	0.40	0.35	
		VEGINT	0.80	0.80	0.88	0.96	0.96	0.96	0.96	0.96	0.88	0.88	0.80	0.80	
		ROOTA	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	
		COIAM	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	

		PCSUCO	30	30	30	30	30	30	30	30	30	30	30	30	
		COLON	60	60	60	60	60	60	60	60	60	60	60	60	
4	High intensity burn area	CAY	0.25	0.25	0.3	0.4	0.4	0.4	0.4	0.4	0.4	0.35	0.3	0.25	
		VEGINT	0.5	0.5	0.55	0.6	0.6	0.6	0.6	0.6	0.55	0.55	0.5	0.5	
		ROOTA	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	
		COIAM	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	
		PCSUCO	10	10	10	10	10	10	10	10	10	10	10	10	
		COLON	30	30	30	30	30	30	30	30	30	30	30	30	
5	Dense aliens	CAY	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	PINE intermediate site prep [ACRU CODE: 6020202] also used in Warburton et al. (2012) for the upper Breede catchment in the Western Cape and labelled Pinus.
		VEGINT	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	
		ROOTA	0.66	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	
		COIAM	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	
		PCSUCO	100	100	100	100	100	100	100	100	100	100	100	100	
		COLON	100	100	100	100	100	100	100	100	100	100	100	100	
6	Scattered aliens	CAY	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	WATTLE [ACRU CODE: 999999991] also used in Warburton et al. (2012) for the upper Breede catchment in the Western Cape and labelled alien vegetation.
		VEGINT	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	
		ROOTA	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	
		COIAM	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	
		PCSUCO	85	85	85	85	85	85	85	85	85	85	85	85	
		COLON	100	100	100	100	100	100	100	100	100	100	100	100	
8	Dryland farming (rooibos)	CAY	0.35	0.35	0.40	0.45	0.45	0.45	0.5	0.6	0.6	0.55	0.5	0.45	All values are set as for Macchia except for VEGINT and CAY, which have been adjusted to represent harvesting from January to March.
		VEGINT	0.5	0.5	0.55	0.72	0.84	0.96	1.08	1.2	1.1	1.1	1	1	
		ROOTA	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	
		COIAM	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
		PCSUCO	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	55.6	
		COLON	100	100	100	100	100	100	100	100	100	100	100	100	
Irrigated areas															
1	Berries	CAYIRR	1	0.75	0.4	0.4	0.6	0.6	0.6	0.6	0.7	0.7	1	1	Based on crop coefficients for blue berries developed by Van der Gulik & Nyvall (2001) divided by 1.2 to account for ET measured from A pan (Schulze 1995) and adjusted to reflect local conditions, growing, irrigation and harvesting crop cycles. COIAIR is set at 0.3 following recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		DINTIR	1.3	1.2	0.9	1	1.1	1.2	1.2	1.2	1.3	1.3	1.3	1.3	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
2	Apples and peaches	CAYIRR	0.5	0.5	0.5	0.5	0.5	0.3	0.3	0.3	0.4	0.5	0.5	0.6	Kc is based on averaging between the two coefficients determined for apples in Ceres and peaches in Wolseley situated in the winter rainfall region in the Western Cape (Gush & Taylor 2014) corrected to equate to A pan equivalent reference evaporation. DINTIR is estimated based on phenology and recommendations in Smithers & Schulze (1995). COIAIR is set at 0.3 following default recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		DINTIR	1.7	1.7	1.7	1.5	1.3	0.7	0.7	0.85	1.25	1.6	1.65	1.7	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
3	Proteas	CAYIRR	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.55	0.5	0.5	

		DINTIR	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.1	1.1	1.1	Derived from Macchia estimates while accounting for irrigation during summer months and minor harvesting impacts. Although CAYIRR considers the plant/soil complex, this is assumed the best available source of information to use to derive estimates due to limited availability of CAYIRR estimates for proteas in the Western Cape. COIAIR is set at 0.3 following recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
4	Olives	CAYIRR	0.5	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.5	0.5	0.5	0.5	Kc determined from Allen et al. (1998) adjusted based on the annual growth cycle of an olive tree in South Africa and corrected by dividing by 1.2 to equate estimates to A pan equivalent reference evaporation (Schulze 1995). COIAIR is set at 0.3 a following default recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		DINTIR	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
5	Buchu or other oils	CAYIRR	0.5	0.55	0.6	0.6	0.6	0.6	0.6	0.5	0.45	0.5	0.5	0.5	Derived from Macchia estimates while accounting for irrigation during summer months, minor harvesting impacts and increased spacing between plants in rows. Although CAYIRR considers the plant/soil complex, this is assumed the best available source of information to use due to limited availability of CAYIRR estimates for buchu in the Western Cape. COIAIR is set at 0.3 following recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		DINTIR	0.7	0.7	0.9	0.9	0.9	0.9	0.9	0.7	0.7	0.7	0.7	0.7	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
6	Tobacco	CAYIRR	0.9	0.4	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.36	0.74	Smithers & Schulze (1995) crop coefficient data for tobacco considering planting date 1 November and DINTIR for tobacco early planting = October 1 [ACRU CODE 3020801] with early planting adjusted based on a November planting date. COIAIR is set at 0.3 following recommendations for irrigated areas from Smithers & Schulze (1995) and Schulze (1995).
		DINTIR	1.5	1	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0	0.5	1	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
7	Artificial pastures	CAYIRR	0.2	0.2	0.4	0.7	0.7	0.7	0.7	0.7	0.8	0.8	0.3	0.2	Pasture annual crop rye grass = April 15 [ACRU CODE 3021002]
		DINTIR	0.5	0.5	0	0.5	0.7	0.8	1	1.2	1.2	0.5	0.5	0.5	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	
8	Tree nuts	CAYIRR	0.65	0.65	0.65	0.65	0.35	0.35	0.35	0.65	0.65	0.65	0.65	0.65	Crop coefficients for Pecans in the Eastern and Western Cape (Smithers & Schulze 1995) and Pecan nuts [ACRU CODE 7770102]
		DINTIR	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	
		COIAIR	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	
9	Annual crops	CAYIRR	0.8	0.7	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.35	0.6	Subsistence crops scattered plants = Nov 1 [ACRU CODE 3040101]. Also, considering typical planting dates for Western Cape.
		DINTIR	1	1	0.6	0.5	0.5	0.5	0.5	0.5	0.5	0	0.5	0.8	
		COIAIR	0.15	0.15	0.2	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.15	
10	Citrus	CAYIRR	0.55	0.6	0.65	0.7	0.65	0.6	0.45	0.45	0.45	0.45	0.45	0.5	Citrus E. and W Cape (Smithers & Schulze 1995) [ACRU CODE 3021102].
		DINTIR	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	
		COIAIR	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	

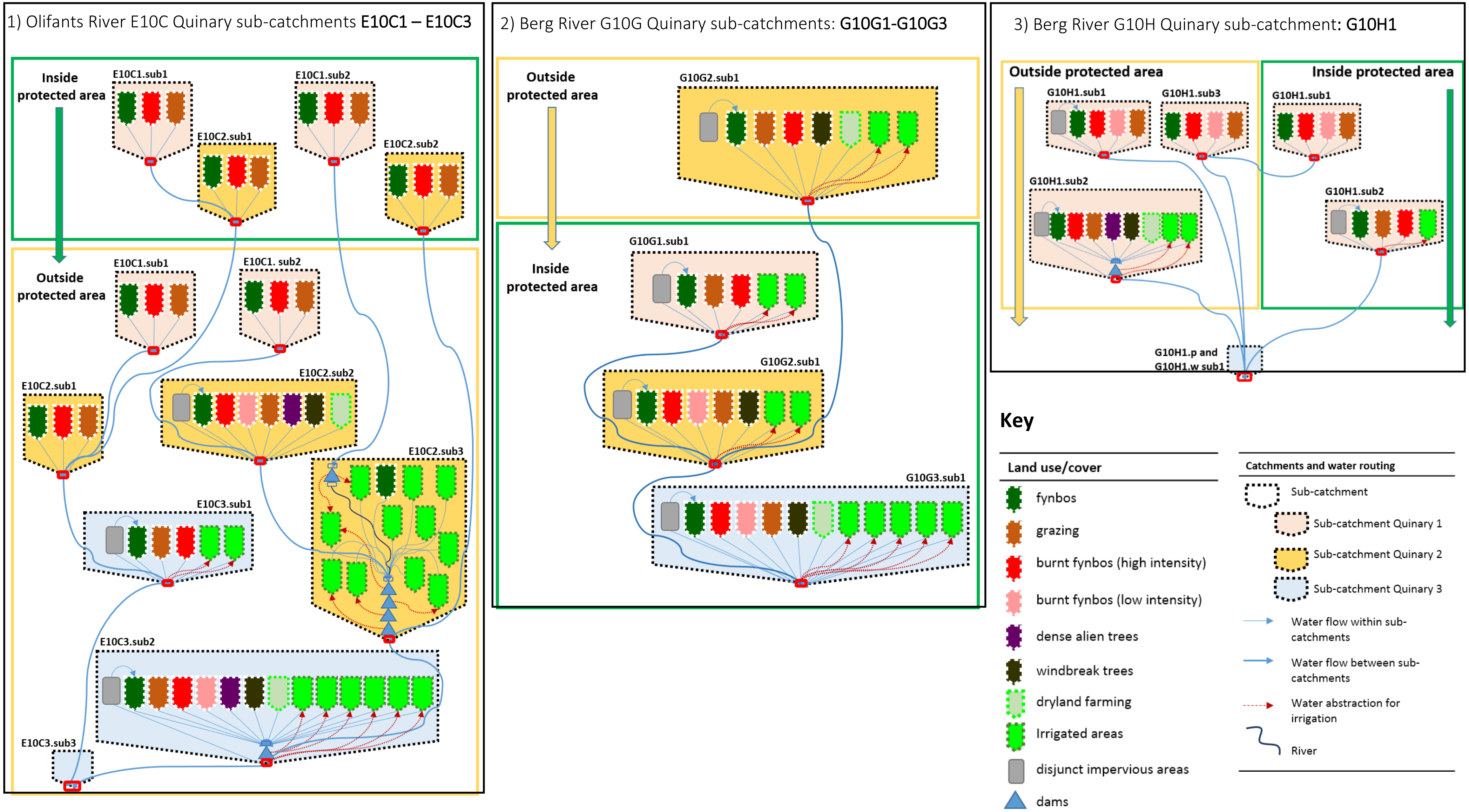


Figure 1 Hydrological model configuration used to model streamflow for outside and inside the protected area for four scenarios 1) 1949, 2) 1972, and 3) 2014 high burn, and 4) 2014 low burn. For further details on hydrological response units, irrigated areas, impervious areas and dams, refer to 5.2.3.3 in Section 5 and Table 1-3 above.

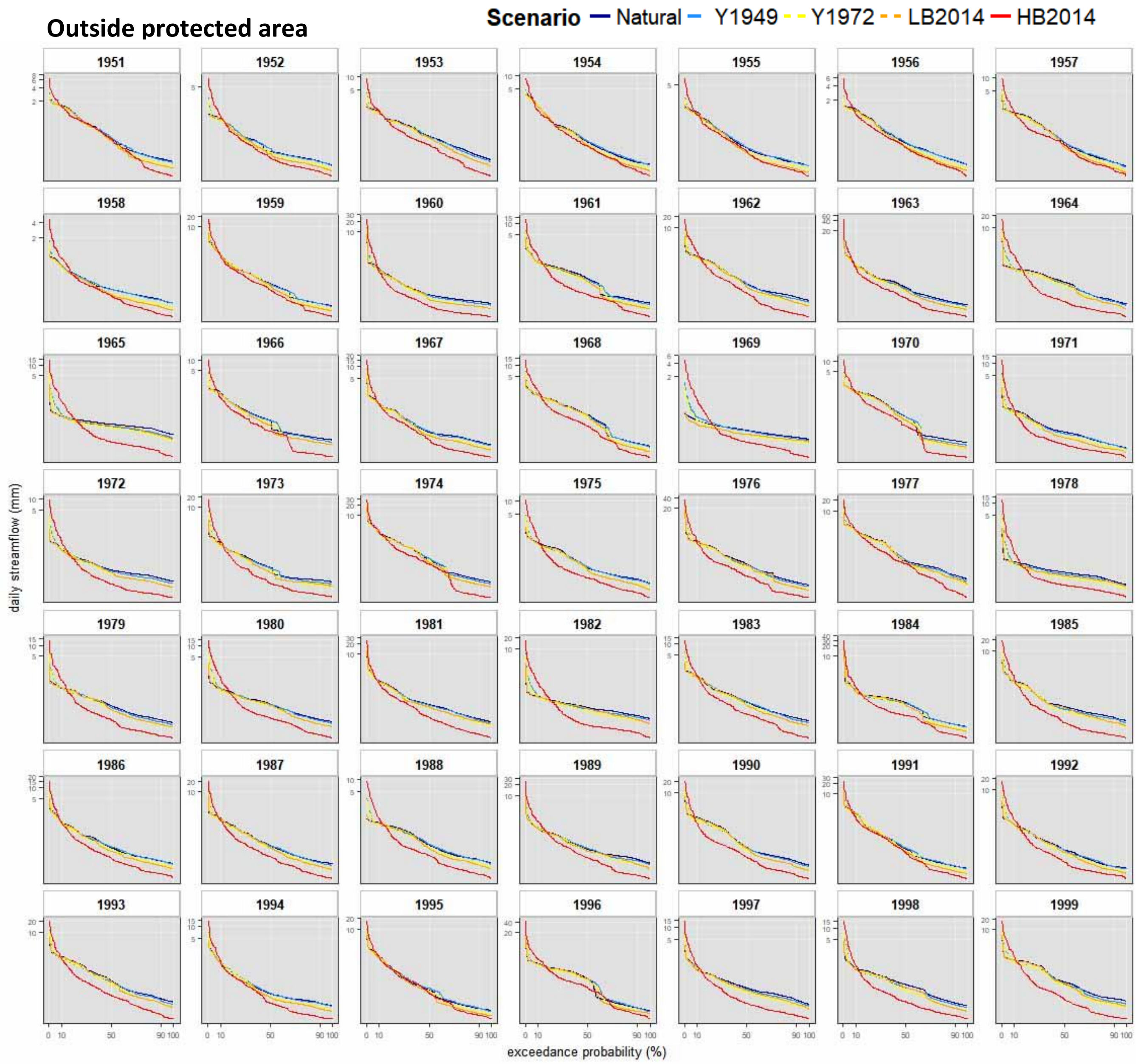


Figure 2 Annual flow duration curves for daily streamflow averaged across 12 sub-catchments for all scenarios modelled outside the protected area. The y-axis is on a log scale.

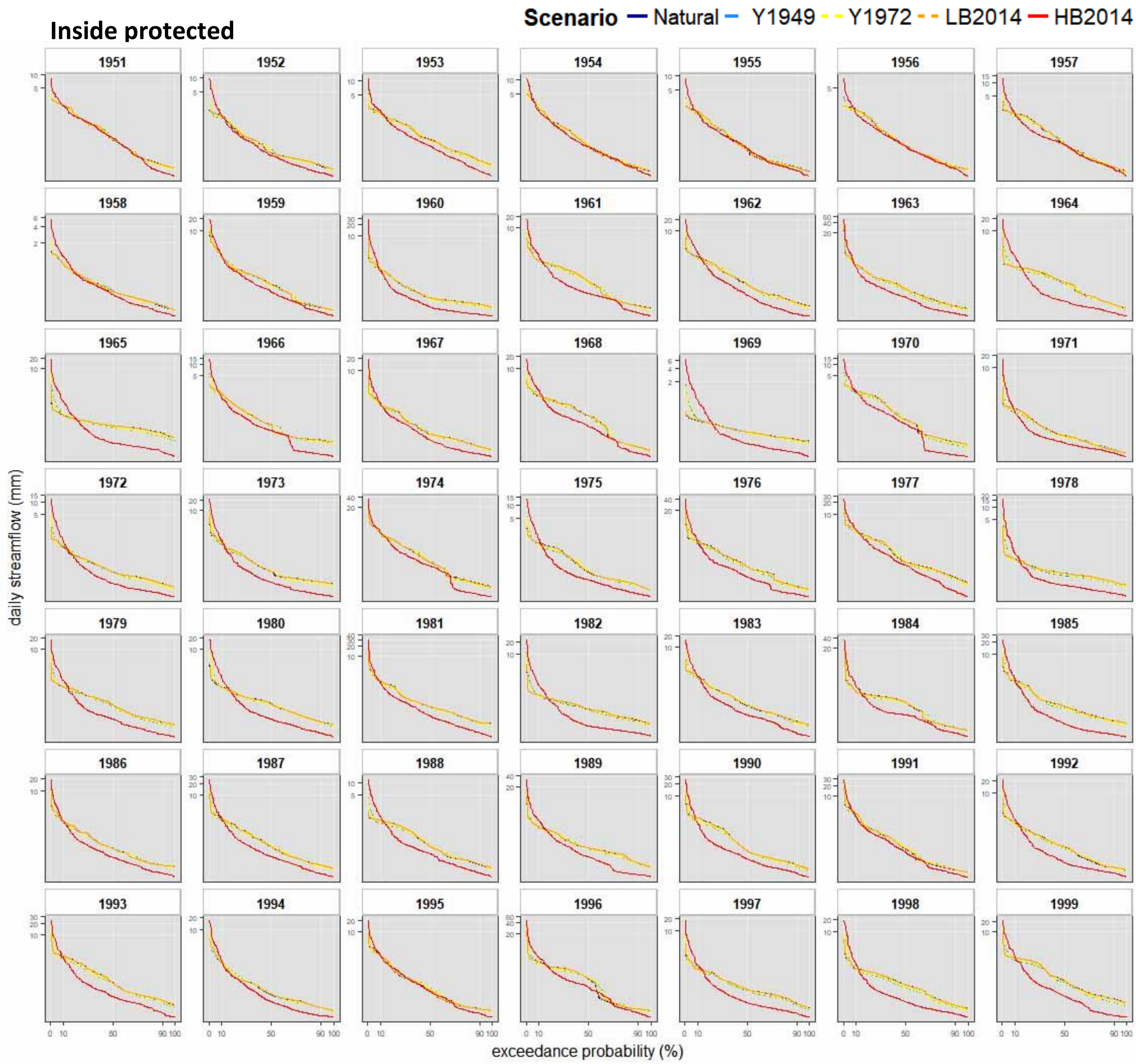


Figure 3 Annual flow duration curves for daily streamflow averaged across nine sub-catchments for all scenarios modelled inside the protected area. The y-axis is on a log scale.

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