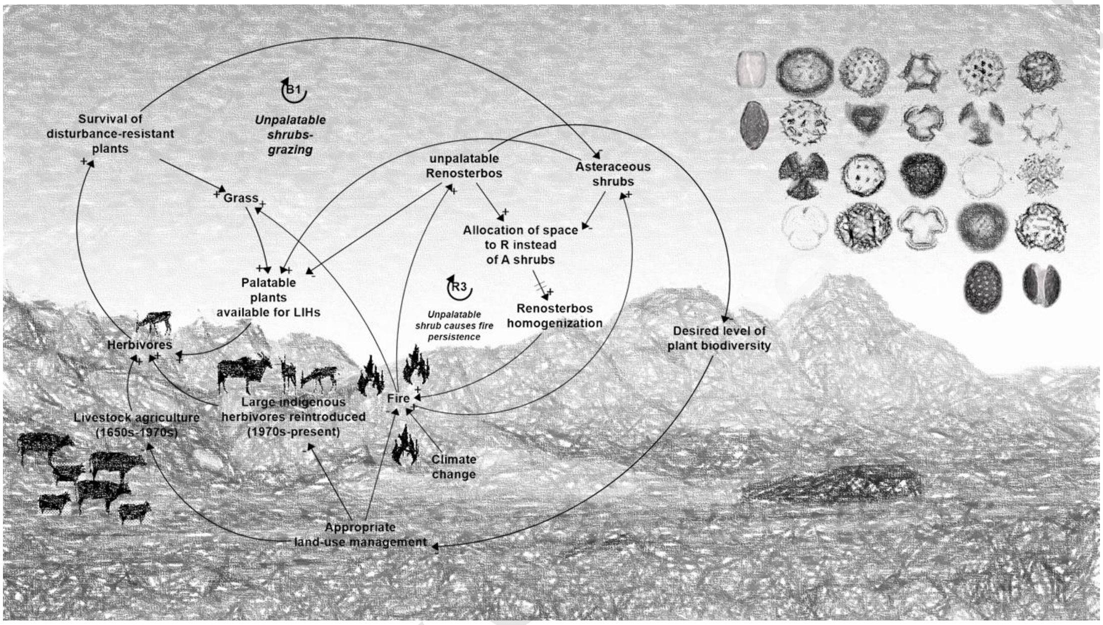
# USING APPLIED PALAEOECOLOGY AND PARTICIPATORY SYSTEM DYNAMICS MODELLING TO INVESTIGATE CHANGES IN ECOSYSTEM SERVICES IN RESPONSE TO CLIMATE AND SOCIAL-ECOLOGICAL DRIVERS WITHIN THE MIDDLE BERG RIVER CATCHMENT, SOUTH AFRICA.





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## DECLARATION

I, Cherié Janine Dirk, 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. The thesis is my own unaided work, both in concept and execution, and that apart from the normal guidance from my supervisor, I have received no assistance except for diatom microscopy and identification by diatom expert Dr Kelly Kirsten.

Signature: Date: 14 November 2021

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*This thesis is dedicated to my best friend and dearest husband, Heinrich Dirk.*

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I am thankful for experiencing the opportunities and challenges conducting this interdisciplinary research. I believe that the power of applied palaeo-data lies in activating this scarce evidence-base by integrating mixed methods to "move beyond research Objective 4". May this be the beginning of me activating change in a sector that I am truly passionate about, and I would like to take this opportunity to thank those who helped me navigate the journey thus far.

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## ABSTRACT

**Research problem:** The Cape Floristic Region (CFR) in South Africa is a globally recognised biodiversity hotspot, which includes conservation and agricultural landscapes that co-produce numerous ecosystem services including biodiversity, water purification, nutrient cycling, invasion resistance, erosion regulation, semi-economic nature-based recreation and ecotourism. Ecosystem services change over time in response to environmental, biotic and social drivers. Because of this a past-present-future lens of environmental change is needed to understand the interactions between these drivers, as a basis for understanding resilience and to inform future scenario planning. Innovative inter- and/or transdisciplinary approaches are needed to improve our understanding of the interacting processes that drive the dynamics of ecosystem service provision in diverse contexts of complex social-ecological systems (SESs). Long-term data from palaeoecology from CFR landscapes can help to understand the historical range of variability in relation to key environmental and anthropogenic drivers. This evidence-based analysis, alongside stakeholder consultation and simulation experiments provides a process-based perspective that can help in the understanding of social-ecological resilience. This in turn can be incorporated into conservation and restoration practices and the sustainable management of ecosystem services.

**Research question:** How can an understanding of the temporal scale and range of ecosystem services be extended through the application of high temporal resolution, multiple-proxy palaeoecological findings? Can the research insights provide land management decision-support to the identification of management thresholds and safe operating spaces for biodiversity conservation?

**Thesis approach and methods:** This study proposed a conceptual meta-framework, the past-present-future lens of environmental change, to explore ecosystem services and drivers of change, stakeholder narratives of change, interactions, feedbacks and dynamic behaviour in the context of sustainable land-use management within multi-functional landscapes of the CFR. Case study sites within the Middle Berg River Catchment focal area were chosen as suitable for palaeoecological analyses. These sites were arranged along an elevation and land-use intensity gradient in order to explore long-term changes in selected ecosystem services (biodiversity, water quality and soil erosion regulation) over decadal-millennial timescales. The sites include an upland conservation site, called Groot Winterhoek Wilderness Area (GWWA), a lowland agricultural site, called Rhenostervlei Farm, and a lowland conservation site, called Elandsberg Private Nature Reserve (Elandsberg PNR). Fossil pollen, charcoal, coprophilous/dung fungal spores, geochemical markers and diatoms were used to investigate changes

in vegetation, fire, herbivory, soil regulation and hydrology over time. The lowland conservation site was then used as a proof of concept for participatory system dynamics (PSD) modelling. This approach used the palaeoecological data as well as stakeholder knowledge to explore trajectories of change under different scenarios of climate, fire and herbivory. Findings from the three case studies were interpreted in relation to conceptual and theoretical frameworks such as complexity-based SESs, systems thinking, mental models, resilience theory (including alternative stable states and thresholds), disturbance and patch dynamics, and non-linearity and closed loop feedbacks.

**Results and Discussion:** Results from the upland conservation site (GWWA) ca. 10 765-1030 BP palaeo-record (GWWA0006) showed a state shift from Drought-Resistant Fynbos shrubland (*Cliffortia*-dominated) and Thicket/Forest mosaic to a Grassy Mesic (ericoid-asteraceous-restioid) Fynbos shrubland and expanding Thicket/Forest mosaic at ca. 4495 yr BP. Changes were driven by increasing moisture caused by climate change after the hot and dry Mid-Holocene Altithermal (MHA). Despite crossing an ecological threshold, the shift in quasi-stable states suggests a resilience to climate change through internal reorganisation of the Fynbos community, while the overall mosaic of Fynbos-Thicket/Forest is maintained. Local and regional fires were abundant and extremely variable over this period. A transitional period from ca. 2885 BP is characterised by unprecedented trends in fire and soil regulation coupled with climate variability. The results suggest ecological character resilience during variable local and regional fire and herbivory regimes over a centennial-millennial timescale. Palaeo-trends of an increase in the incidence of local fire and *Cliffortia* abundance during the warmer Medieval Climate Anomaly (MCA) is congruent with stakeholders' narratives of recent change since ca. AD 1970s, suggesting changes in *Cliffortia* abundance and increased fire might be expected as the climate becomes warmer and drier in future.

Results from the lowland agricultural site (Rhenostervlei Farm) ca. AD 1795-2011 palaeo-record (RV3) showed Renosterveld and Fynbos shrubland (i.e. *Elytropappus rhinocerotis* and other Asteraceae species) and Thicket (*Oleaceae* and *Searsia*) mosaic elements are present in the landscape. Their presence is associated with low to moderate erosion and land-use disturbance which led to an initial drying out period and high nutrient water quality. This changed slightly to wetter, brackish-saline, medium to high nutrient water quality until ca. AD 1960. Thereafter, there was a decrease in Renosterveld and Fynbos shrubland patches and a contraction of Thicket patches. This state transition is associated with a dramatic increase in invasive alien plants (Eucalyptus) due to local and regional agricultural intensification including vegetation clearing for crop and pasture production and increased fire and herbivory disturbance. General ecosystem services remained stable, however, despite intensive land-use disturbance coupled with climate variability. This suggests only partial ecological degradation as Rhenostervlei Farm has retained resilience of ecosystem services through the preservation of key functional traits and structure of the vegetation despite the introduction of novel ecological components. Narratives of change from the stakeholders show an informed understanding of the influence of both indigenous and alien trees, the need for proactive fire management and river rehabilitation to improve plant biodiversity, water quality and soil erosion regulation, especially in the lowlands.

Results from the lowland conservation site (Elandsberg PNR) used the ca. AD 750-2012 palaeo-trends published by Forbes et al., (2018) as the basis for a simulation experiment using a PSD modelling approach. The Ecological Model simulated the transition from Asteraceae long-spine type-1 pollen to Asteraceae Stoebe/Elytropappus-type pollen coupled with increased coprophilous fungal spores and macro-charcoal. The dynamic hypothesis articulated a state shift in causal loop dominance from multiple Asteraceous shrubs abundance to unpalatable *Elytropappus rhinocerotis* (Renosterbos). Output from the Ecological Model replicates the palaeo-record that suggests that from the ca. AD 1800s the main changes in the environment were driven by grazing and that an ecological threshold was crossed in ca. AD 1950s coupled with an unprecedently high local fire regime. This state transition was due to agricultural intensification and these trends continued through the conservation period from ca. AD 1970s to the present when large indigenous herbivores were reintroduced into the landscape. The Ecological Model's future scenario analysis suggests that it may be difficult to return the ecological system to its pre-1950s historical range of variability even if grazing and fire intensity are reduced. Therefore, hysteretic system behaviour must be considered when setting management thresholds for adaptive grazing-fire management. As well as reducing fire and grazing, manual clearing of unpalatable Renosterbos is recommended to decrease the likelihood of a future regime shift to a degraded alternative stable state that is undesirably resilient due to its resistance to change or reversibility to a desirable state.

**Synthesis:** The primary contribution of this study is in providing a SES research case study that illustrates how the evidence-based and descriptive palaeoecological data can be applied in a more process-based and prescriptive way. Specifically, long-term data allows the effects of interacting drivers on ecosystem services to be explored. These in turn provide reference conditions/states, which can provide a basis for exploring a process-based understanding of change over time. This can be achieved via participatory future scenarios planning and can contribute to defining management thresholds with stakeholders. The palaeo-records from the upland and lowland sites suggest overall resilience and maintenance of ecosystem function despite climate change, increasing anthropogenic disturbance and loss of plant biodiversity. The simulation experiment from Elandsberg PNR's proof of concept shows that the ecosystem can lead to a hysteretic collapse due to Renosterbos homogenisation. The maintenance of ecosystem function and ecosystem services use in the future requires further qualitative and quantitative PSD analyses as it is likely that critical ecological thresholds are imminent, and managers need to set thresholds and targets accordingly.

Useful end-products are potential leverage points for informing restoration of ecosystem function and SES resilience. This approach can enhance mainstreaming long-term data into multiple environmental sectors. The interactive simulation model interface (Land Management Decision-Support Tool) translates long-term data that can be used by policymakers and land-use managers. It helps to generate insights on how to maintain ecological integrity that preserves supporting/provisioning and regulating services which, in turn, benefit complex SESs. Furthermore, the proposed conceptual meta-framework could be used to identify the common ecological traits and environmental conditions across multiple sites within the CFR. With such information, ecosystem services indices can be quantified, and indicators of resilience can be developed with the benefit of communicating a broad-scale overview of environmental change during future ecosystem assessments.

**Broader significance:** Quantifying ecosystem services requires the comparison of social-ecological parameters to reference conditions and states. This can be a challenging concept when long-term data is lacking for disturbance-driven ecosystems of the CFR. However, this study demonstrates how applying a past-present-future lens of environmental change over time can improve our understanding of temporal variability and dynamic feedbacks between ecological parameters. The changes in ecosystem services and drivers that influence ecological and structural character resilience, the presence of functional redundancy, systems delays and non-linearity that govern desired resilience, or the lack thereof, are analysed in this study. Identifying the historical range of variability is essential in defining reference conditions and in informing current and future biodiversity conservation in multi-functional landscapes. It also highlights the identification of potential management thresholds and targets that can be further quantified via calibration with modern analogues in future research.

**Keywords:** *ecosystem services, land-use, state transitions/regime shifts, thresholds, resilience, mixed methods, multiple-palaeoecological proxies, multi-stakeholder engagement, system dynamics modelling, complexity-based socio-ecological systems, inter- and transdisciplinary*

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## GLOSSARY OF TERMS

Given the interdisciplinary nature of the research, the intention of this glossary is to assist the reader access multiple disciplinary terminologies. It is intended as a guide as opposed to being definitive.

| | |
|---|---|
| Adaptive capacity | Capacity of social–ecological systems, including both their human and ecological components, to respond to, create and shape variability and change in the state of the system. (Folke et al., 2002). |

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| Adaptation readiness | Identifying and characterizing what is actually being done to prepare for adaptation, focusing on the strength and existence of governance structures that determine the preparedness to build support for adaptation action and effectively develop, implement, and monitor adaptation interventions (Ford and King 2015). |
| Community of Practice | A group of people attending to a particular knowledge area. A community of practice aims to produce knowledge fit for end-users, without the constraints of settled scientific paradigms (Buschke et al., 2019). |
| Boundary crossing | A generative process that involves the effective integration of diverse knowledge types and domains while creating new knowledge through collaborative networks and processes (Jean et al., 2018). In the learning sciences it is defined as the interactions and transitions between agencies and stakeholder groups across different domains and boundaries (Akkerman and Bruining, 2016), including information flow between science providers and science users through decision-support and science-delivery methods and frameworks. In this study, this term does *not* refer to the transition between biome/vegetation boundaries or ecotones and disciplinary boundary crossing is also promoted within the interdisciplinary research approach. |
| Boundary object | A boundary object is information, such as specimens, artefacts, maps, etc. which can be used in different ways by different communities to facilitate dialogue, engagement and learning amongst multiple stakeholders (Fischer and Riechers, 2019; Star and Griesemer, 1989). |
| Ecosystem-based Management | Ecosystem-based Management "describes the comprehensive integrated management of human activities based on the best available scientific knowledge to achieve sustainable use of ecosystem goods and services and maintenance of ecosystem integrity" (O'Higgins et al., 2020). |
| Ecosystem services | The benefits that society derives from nature/ecosystems. There are four broad categories of ecosystem services: supporting, |

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|---|---|
| | provisioning, regulating and cultural (Millennium Ecosystem Assessment, 2005). |
| Ecosystem stewardship | A strategy to respond to and shape social–ecological systems under conditions of uncertainty and change to sustain the supply and opportunities for use of ecosystem services to support human well-being (Chapin et al., 2010). Human well-being: quality of life in terms of material needs, freedom and choice, good social relations and personal security. |
| Mental models | A mental model of a dynamic system is a relatively enduring and accessible, but limited, internal conceptual representation of an external system (historical, existing or projected) whose structure is analogous to the perceived structure of that system (Doyle and Ford, 1998, 1999). |
| Past-present-future continuum | A long-term perspective employed in research design and implementation – e.g. Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014. |
| Past-present-future lens of environmental change | The conceptual meta-framework as defined in the present study uses mixed methods to explore ecosystem services and drivers of change in the past (high resolution, multi-proxy palaeoecological data), present (stakeholder perceptions) and future (system dynamics qualitative and quantitative modelling results of future scenarios). |
| Quasi-stable state | A more easily reversable regime that is quasi-stable since it remains stable for a significant length of time without crossing a critical ecological threshold (Gillson et al., 2020; Todman et al., 2016). |
| Resilience | Capacity of a social–ecological system to absorb a spectrum of shocks or perturbations and to sustain and develop its fundamental function, structure, identity and feedbacks as a result of recovery or reorganization in a new context. Resilience is the ability to withstand disturbance without loss of critical functions and/or the ability to recover from disturbance (Carpenter et al., 2001; Chapin et al., 2010; Folke, 2006; Folke et al., 2004; Holling, 1973; Lew et al., 2016; Marchese et al., 2018; Quinlan et al., 2016; Walker and Meyers, 2004). |

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|---|---|
| Resistance or undesirable/negative resilience | The ease or difficulty of changing the system; how resistant it is to being changed (Lake, 2013; Oliver et al., 2015). |
| Social-ecological systems(s) | A complex adaptive system with strong connections and feedbacks within and between social and ecological components that determine their overall dynamics (Biggs et al., 2021). |
| Sub-zones | Not statistically significant zonation in the palaeo-record that were identified using the cluster analysis. Sub-zones showed subtle changes important for interpreting data according to Objectives 2 and 3. As defined in the present study. |
| Sustainability | Use of the environment and resources to meet the needs of the present without compromising the ability of future generations to meet their needs (Chapin et al., 2010). |
| Table Host | A co-facilitator trained to host and mediate the dialogues during the multi-stakeholder engagement workshop which was an adaptation of participatory modelling to elicit variables. The Table Host is required to be impartial and neutral, while mediating the dialogue to encourage fair contribution amongst participants and capturing feedback in a transparent way. As defined in the present study. |
| Zones | Statistically significant zonation of pollen and diatom records based on cluster analyses. This is to show the relationship between plant biodiversity and water quality and drivers of change. As defined in the present study. |

**LIST OF ACRONYMS**

| | |
|---|---|
| CFR | Cape Floristic Region |
| CLD | Causal Loop Diagram |
| CoP | Community of Practice |
| EbA | Ecosystem-based Adaptation |
| ES | Ecosystem Services |
| FDI | Fire Danger Index |
| GMB | Group Model Building |
| GWWA | Groot Winterhoek Wilderness Area |

| | |
|---|---|
| IAPs | Invasive alien plants |
| LAC | Limits of acceptable change |
| LIA | Little Ice Age |
| LIHs | Large indigenous herbivores |
| LoI | Loss on Ignition |
| MAP | Mean Annual Precipitation |
| MCA | Medieval Climate Anomaly |
| M&E | Monitoring and Evaluation |
| MS | Magnetic Susceptibility |
| MHA | Mid-Holocene Altithermal |
| PNR | Private Nature Reserve |
| PSD | Participatory System Dynamics |
| SD | System Dynamics |
| SES | Social-ecological system |
| SFD | Stock and flow diagram |
| TPC | Threshold of Potential Concern |
| UCT | University of Cape Town |
| VANG | Vangkraal Spring sediment core |
| VCT | Validation cessation threshold |
| WWF | World Wildlife Foundation |
| WRZ | Winter Rainfall Zone |

# 1 CHAPTER 1: GENERAL INTRODUCTION

## 1.1 Problem Statement: Managing ecosystem services within the Cape Floristic Region (CFR)

The general context of this study is based on the concept of ecosystem services, which are the benefits that nature provides to people and on which all humankind depends. There are four broad categories of ecosystem services: supporting, provisioning, regulating and cultural (Millennium Ecosystem Assessment, 2005). Ecosystem services come from ecological infrastructure (Cumming et al., 2014) – e.g. healthy mountain catchments, wetlands, corridors of natural habitat, which together form a network of interconnected structural elements in the landscape. The Cape Floristic Region (CFR) (Figure 1.1) is one of the world's 25 biodiversity hotspots (Goldblatt and Manning, 2000; Low and Rebelo, 1996; Mittermeier et al., 1998; Myers et al., 2000), and contains >9000 plant species, of which ca. 68% are endemic (Manning and Goldblatt, 2012). The CFR comprises multi-functional landscapes, consisting of conservation, agricultural and urban land. As conservation and agricultural landscapes co-produce (Fischer and Eastwood, 2016) and depend on ecosystem services (e.g., biodiversity, water purification, nutrient cycling, invasion resistance, erosion regulation), they play a critical role in food and water security, economic development and overall resilience to threats from climate change.

## 1.2 The Research Problem: How variability in ecosystem services affects sustainable land-use management in the CFR

Human impact threatens the CFR’s unique plant biodiversity with about 30% of indigenous CFR vegetation already transformed (Myers et al., 2000), making it a global conservation priority. The Mediterranean-type ecosystem of the CFR consists of four biomes, one of which is the Fynbos biome. Vegetation within the Fynbos biome is classified as evergreen, hard-leaved, fire-adapted shrubland or heathland and commonly occurs on nutrient-poor soils (Bergh et al., 2014; Moll and Jarman, 1984; Van Wilgen, B., Richardson, 1985). The name of this sclerophyllous vegetation is derived from the Old Dutch and Afrikaans terms for 'fine' and 'bush', reflecting the finely-divided and bushy nature of the vegetation, Two dominant vegetation types within the Fynbos biome are Fynbos and Renosterveld and are the focal vegetation types in this study (see descriptions in sections 3.1.1 and 3.2). Current threats include urban development, agriculture, the spread of invasive alien plants (IAPs) and anthropogenic climate change (Allsopp et al., 2014a; Goldblatt and Manning, 2000; Myers et al., 2000). The CFR has been impacted by climate variability and humans for millennia. People have manipulated fynbos landscapes for millennia. Firstly, San hunter-gatherers (from 10 000 BP) and later Khoikhoi pastoralists (from 2 000 BP) (Cooke, 1965; Deacon, 1992; Elphick, 1977; Humphreys et al., 1998; Westphal, 1963)intentionally burnt the vegetation while the arrival of European settlers in the 17th century brought an intensification of agricultural activities especially from the 20th century due to technology and industrialisation. Increased transformation in recent decades has potentially impacted on ecosystem services, and uncertainty related to future climate change projections has increased pressure on sustainable land-use management for both conservation and agriculture. Long-term information on changes in ecosystem services is lacking, and we, therefore, do not know whether trends in recent decades are unprecedented, still within the historical range of variability or approaching critical ecological thresholds. An understanding of long-term changes in ecosystem services in response to multiple interacting drivers could help to conserve ecosystem services and manage them sustainably.

## 1.3 Rationale and approach

### 1.3.1 Why a past-present-future lens of environmental change is needed

To address the research problem and explore innovative approaches and practices in the field of palaeoecology, I argue that combining mixed methods such as palaeoecology and participatory system dynamics (PSD) and embedding these methods in a past-present-future continuum (Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014) helps to understand trajectories of change and assist in anticipating future change and managing sustainably. Social-ecological systems (SESs) research often requires the adaption or sequential implementation of methods that capture both social and ecological domains, as well as their dynamic interdependencies (de Vos et al., 2019). Therefore, SESs research and practice will benefit from a more contextual (evidence- and process-based) understanding of long-term change by adopting the conceptual meta-framework in this study: a past-present-future lens of environmental change.

#### *1.3.1.1 The need for applied palaeoecology*

Though the need to understand the processes that drive landscape change is well-recognised, there is still little direct application of palaeoecology in conservation and sustainability science. John Birks, who has influenced the advancement of the applied and quantitative palaeoecology suggested 25 years ago that applied palaeoecology can make a significant contribution to conservation (Birks, 1996). Over the years, studies in various parts of the world have shown this to be true by demonstrating the importance of palaeoecological research for informing biodiversity conservation and land and water management (Birks, 2012; Dearing, 2008; Gell et al., 2005; Gell, 2012; Gell et al., 2018; Gil-Romera et al., 2010; Gillson and Willis, 2004; Pederson et al., 2006; Willis and Birks, 2006). Applied palaeoecology describes the role of long-term data to increase our understanding of changes in ecosystem services, ecosystem resilience and variability and incorporate insights in ecosystem assessments and management (Dearing et al., 2012; Gillson, 2015; Gillson and Marchant, 2014; Jeffers et al., 2015).

A long-term palaeo-perspective on environmental change provides improved evidence of historical variability and a unique opportunity to identify the interacting drivers of environmental change caused by climate change and land-use disturbance. Despite global efforts, Jeffers et al. (2015) reiterate that the applied palaeoecological community of practice (CoP) must take the lead in demonstrating the benefits derived from including palaeo-data within existing ecosystem management and research, including ecosystem-based management approaches (Brink et al., 2016; O'Higgins et al., 2020; Schoeman et al., 2014). Although a few South African-based applied palaeoecological studies have successfully evaluated the extent and nature of changes in the landscape and developed management insights for conservation and restoration (Dabengwa et al., 2021; Forbes et al., 2018; Gillson and Duffin, 2007; MacPherson et al., 2019), what more can be done to operationalise and mainstream palaeoecology for sustainable development? What conceptual frameworks, theories and approaches/methods are needed to ensure that applied palaeoecology actively assists land management and conservation efforts in the context of CFR landscapes and societies?

##### *1.3.1.2 The need for systems thinking and stakeholder engagement to understand complex SESs*

The embedding of a past-present-future continuum (Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014) within systems thinking can provide a novel perspective on sustainability within the CFR. Systems thinking offers a broad perspective that uses knowledge, tools, and techniques to understand change and unravel complexity by evaluating overall structure, patterns and feedbacks. While systems thinking allows a process-based understanding of SESs, it also opens the door to understanding how systems change over time when the interactions between driving factors and responding factors change across multiple spatial and temporal scales (Biggs et al., 2021; Folke et al., 2002; Folke, 2006; Maani and Cavana, 2007; McGinnis and Ostrom, 2014; Ostrom, 2009; Preiser et al., 2021; Richmond, 1993). SESs do not respond in an incremental and predictable way to increasing or decreasing external pressures, an effect is rarely proportional to the cause, and what happens locally in a system often does not apply in distant regions (other states of the system or patches in the landscape) (Reyers et al., 2018; Sterman, 2002). Therefore, complexity-based SESs are non-linear in their dynamics. Systems thinking is, therefore, an effective broad approach that provides a rationale for applying systemic and interdisciplinary methods (e.g. applied palaeoecology and participatory system dynamics) to explore resilience, thresholds, tipping points, collapse and recovery and understand dynamic complexity within SESs (Gunderson and Holling, 2002; Hossain et al., 2017; Johnson, 2013; Young et al., 2019). These concepts are featured in the literature review (Chapter 2) and later discussed in Chapters 5-8.

One aspect of complexity-based SESs is the plurality of perspectives and the role of stakeholder perceptions including beliefs, values and assumptions. Non-linearity also arises as multiple factors interact in decision-making. Understanding this requires that researchers engage with stakeholders from different sectors of society. Stakeholder engagement occurs on a spectrum of stakeholder participation. A relatively low level of participation entails little more than consultation with stakeholders while a high level of stakeholder participation would include collaboration or shared learning between researchers and stakeholders who are decision-makers that represent government, private, and civil society groups.

To achieve sustainable ecosystem management, the process by which information is derived and utilised needs to include stakeholder participation and mutual learning (Knight et al., 2008; Wheeler et al., 2019). Participation from multi-stakeholders creates an opportunity for social learning as it reinforces the articulation and evolution of shared desirable visions, goals, and community capacity development for the future (Helfgott, 2018; Roux et al., 2017; van den Belt and Blake, 2015; Videira et al., 2010; Voinov and Bousquet, 2010). To understand the social-ecological context, and develop end-products that are useful, expert input is needed to inform the development of a systems model based on real-world problems.

Models can serve as boundary objects (Fischer and Riechers, 2019; Star and Griesemer, 1989) that focus the attention of stakeholders on systems of common interest. There are various types of stakeholder-based modelling approaches (e.g. participatory modelling, group model building, mediated modelling, collaborative learning), all with their associated software and analytical tools (Voinov and Bousquet, 2010). Despite the challenges of often conflicting interests and blurred boundaries in spatial, social and ecological contexts, stakeholder engagement is common and best practice systems thinking and modelling methodologies at present. It has proven to be useful in environmental assessment and decision-making processes, particularly in supporting problem scoping and policy analyses (Bou Nassar et al., 2020; Inam et al., 2015; Langsdale, 2007; Videira et al., 2010; Voinov and Bousquet, 2010).

Engagement with key stakeholders from multiple sectors has also been effective in conservation science in the CFR (Balmford, 2003; Cowling and Pressey, 2003; Gelderblom et al., 2003; Rouget et al., 2014). As an applied palaeoecological Community of Practice (CoP) there is an urgent need to communicate research results in such a way that they can be incorporated into the practice of land-use management in the region. In the present study, stakeholder engagement allows for reflection on past long-term dynamics of ecosystem services and drivers. It also provides insights into the connections and relationships underlying persistent sustainability problems which need to be addressed in the present and in the future. Furthermore, consideration of multi-stakeholder perceptions allows for transparency and inclusivity and is a form of model validation that deals with a meta- and contextual level of complexity. It sets out the purpose and boundary conditions of SES problems, and can be executed without computational support (Groesser and Schwaninger, 2012). Research has shown that links between different stakeholders empowers and motivates resource users to conserve nature and use natural resources wisely (Gelderblom et al., 2003). This begs the question, do all land-use managers (e.g. commercial farmers, conservation practitioners and government authorities) perceive that current policies and their management measures have a positive impact on the conservation of biodiversity and the integrity of ecosystem services and ecosystem function? Do multiple stakeholders value the land management decision-support derived from an interdisciplinary study such as this one which is related to the management of plant biodiversity, water, erosion, fire and grazing?

#### *1.3.1.3 The need for systemic approaches to enhance our understanding of complex SES - System Dynamics (SD) modelling as a tool*

System dynamics (SD) modelling, as a subset of systems thinking, is one of the many approaches that offers effective tools for better understanding complexity and social-ecological dynamics as it focuses on the interrelatedness of causes and effects of a complex problem. SD requires a comprehensive and genuine inquiry to explore how behaviours might change over time as a consequence of shifts in the relative strengths of the underlying feedback loops in a system (Maani and Cavana, 2007; Richmond, 1993). Furthermore, participatory system dynamics (PSD) as a nuanced approach applies the use of system diagrams and computer simulation models in group settings (Kopainsky et al., 2017; Voinov and Bousquet, 2010). It includes a mediated process that allows for mental model refinement, alignment, and the creation of a shared language amongst multiple stakeholders (see detailed explanation in section 2.3 and 2.4). In addition to prediction and forecasting, there are different reasons and motivations for developing a SD model (Epstein, 2008), which are relevant in the local and international environmental management sector (Ford, 2000). Several studies demonstrate the application of SD in environmental management interventions, such as water resource management (Beall et al., 2011; Clifford-Holmes et al., 2017a, 2017b; Rehan et al., 2011; Simonovic, 2009; Stave, 2003; Tidwell et al., 2004; Wang et al., 2011; Winz et al., 2009), marine ecosystems (Weller et al., 2016, 2014), fire management (Collins et al., 2013), conservation agriculture (B. L. Turner et al., 2016; Turner and Kodali, 2020; Von Loeper et al., 2016), ecological restoration (Crookes et al., 2013; Menendez et al., 2020; B. L. Turner et al., 2016) food systems (Bennich et al., 2018; Brzezina et al., 2017, 2016; Herrera de Leon and Kopainsky, 2019; Stave and Kopainsky, 2015), resilience (Bennett et al., 2005; Ciobanu and Saysel, 2020) and scenario planning (Allington et al., 2018; Miller et al., 2017; K. G. Turner et al., 2016). There are meaningful SD applications in climate change adaptation and decision-support tools in South Africa (Brent et al., 2017a, 2017b; Carnohan et al., 2021), which provide motivation for the present study to add to this sum of knowledge. This qualitative and quantitative approach of combining palaeoecological data on plant biodiversity, herbivory and fire history with a SD model is a rare and novel contribution to sustainable land-use management in the CFR, South Africa, and the broader African continent.

### 1.3.2 The interface between palaeoecology and modelling

An intriguing interface exists between palaeoecology and modelling, which is grounded by the usability of scientific information. The evidence-based and descriptive knowledge generated from applied palaeoecological research needs to be translated into formats that are useful to stakeholders (see section 4.2.1). This increases the value of end-products and thus the likelihood that they will be used by decision-makers within designed and/or self-organised systemic structures, such as existing governance structures in multiple sectors. Such meaningful end-products can then be used as a boundary object to facilitate dialogue amongst multiple stakeholders (Fischer and Riechers, 2019; Star and Griesemer, 1989). A boundary object is information, such as specimens, artefacts, maps, which can be used in different ways by different communities (e.g. commercial farmers, conservation practitioners, land-use governance authorities) in engagement and learning. Multiple stakeholders may interpret the boundary object differently, but it contains enough immutable content to maintain its integrity. Boundary crossing can occur when the model facilitates communication between different stakeholders by making explicit the knowledge and assumptions that emerge during the development and/or interpretation of the model. In this format, the palaeo- and model outputs can be used as a boundary object to facilitate the participatory planning of future scenarios. What do we know from the palaeo-record of the past, and what are the current systemic structures and value-systems that govern the changes in SES? With this new contextual knowledge, how can our evidence- and process-based understanding help us plan for multiple future scenarios in a changing and uncertain world?

### 1.3.3 The study focal area

The current and future resilience and sustainability of ecosystem services are influenced by landscape history of the CFR. A range of drivers affect processes and states of ecosystems. However, there is limited information on the long-term changes in ecosystem services and drivers, and therefore it is difficult to assess whether recent changes in land-cover in the CFR are sustainable. The Berg River Catchment (Figure 1.1) provides an opportunity to investigate the benefits of having a longer-term past and future perspective on ecosystem services, to build more contextual arguments for sustainable land-use management within the region. The selection of three study sites along a gradient of elevation and land-use intensity within the Middle Berg River Catchment allows for a deep exploration of the relative roles of anthropogenic and environmental factors both spatially and temporally. Refer to the Study Focal Area Chapter 3, section 3.2 which provides further justification for the selection of each site. The relatively high elevation site, Groot Winterhoek Wilderness Area (GWWA), represents the lowest level of anthropogenic impact, the mid-elevation site, Elandsberg Private Nature Reserve (Elandsberg PNR), represents intermediate impact, with some historical agricultural use, and the relatively low-elevation site, Rhenostervlei Farm, provides an example of intensive land-use (Figure 1.1 and Chapter 3). The agricultural lowlands are equally important for studying changes in ecosystem services in comparison to conservation sites due to the "co-production" of ecosystem services - a term which refers to the interactions between people and place that lead to ecosystem services (Bengtsson et al., 2021, 2003; Fischer and Eastwood, 2016). People need to be considered as an integral feature of the system as their identity and capabilities add to the benefits generated, especially cultural and provisioning services

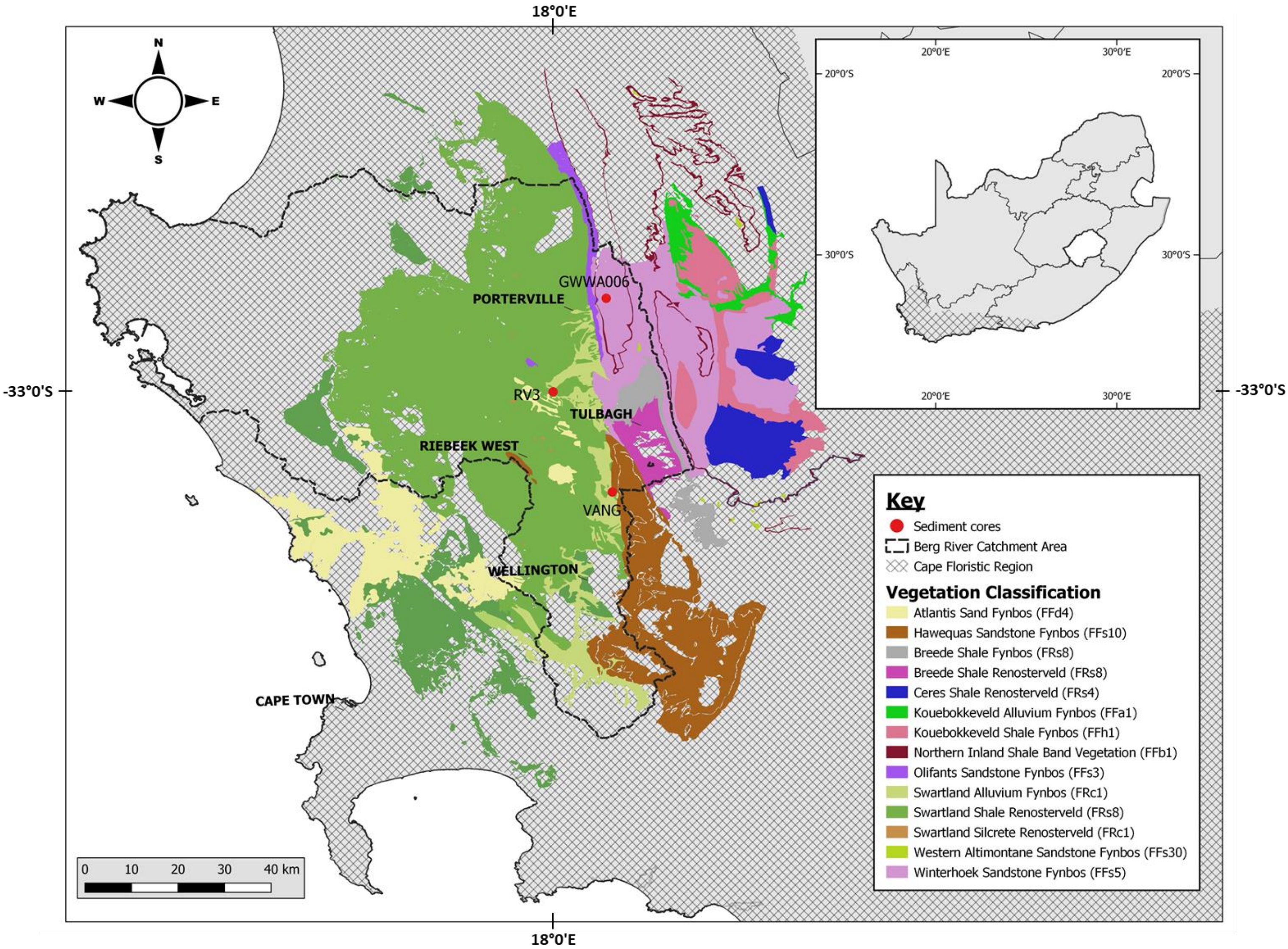



***Figure 1.1:** Map showing the general study focal area, Berg River Catchment covering various vegetation types in the Cape Floristic Region, South Africa. Three sediment coring sites are highlighted: GWWA006 from Groot Winterhoek Wilderness Area, VANG from Elandsberg PNR and RV3 from Rhenostervlei Farm.*

## 1.4 Project Aim and Objectives: Using the past-present-future lens of environmental change as a conceptual meta-framework for exploring changes in ecosystem services over time

The overall aim of the present study is to provide a past-present-future lens of environmental change in the study area, which provides information that could contribute to sustainable management of ecosystem services. The research project was designed around four key objectives and three components summarised in Figure 1.2 with research questions and hypotheses for Objectives 1-4 and

Component B detailed in Table 1.1.Component A and Component C are supporting components for Objectives 1-4 and therefore do not have associated research questions. This conceptual meta-framework shows how long-term records from palaeoecology can be interpreted in terms of changes in ecosystem services resulting from drivers caused by land-use and climate change. Palaeoecology has the unique position to provide insights on historical variability, giving clues on climatic and anthropogenic-induced impacts over decadal-millennial timescales. This temporal understanding of change can be interpreted in terms of landscape processes and interactions, while focusing on the combination of feedbacks (from systems thinking and SD models) that occur across multiple temporal and spatial scales. This methodological approach encourages long-term reflection and allows for participatory exploratory scenario development through co-creation and co-learning. The approach promotes the holistic exploration of system interactions including the connections between ecosystem services and drivers that threaten ecosystem integrity at multiple levels of complexity. It advances the practical implications of applying the past-present-future continuum proposed by previous studies (Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014). Therefore, a past-present-future continuum is adapted to help us better understand environmental change and research objectives are further positioned within an integrative framework adapted from Dearing et al., (2015, 2012) (Table 1.1) to synthesise insights of the complex SESs. The mixed methods approach leads to a deeper understanding of complex adaptive systems, motivated by the aim of integrating long-term data into more effective decision-support for land-use managers within the CFR.

### 1.4.1 Palaeoecology to provide insights on the *past* (Objective 1-3)

There is limited long-term palaeoecological data to adequately evaluate the extent and nature of changes in the CFR's multi-functional landscapes. High temporal resolution, multi-proxy palaeoecological studies facilitate researchers to cross boundaries between neoecology (i.e. typically widely used ecological field data collection techniques (Clements et al., 2021)) and palaeoecology within SES research. The study utilises multiple palaeo-proxies including fossil pollen, diatoms, spores, charcoal, sediment properties and geochemical markers (section 4.1) to reconstruct environmental history at two sites, GWWA and Rhenostervlei Farm, within the Middle Berg River Catchment area over the past ca. 215 to 10 000 years, and to explore the drivers of these changes (Objectives 1 and 2, Figure 1.2 and Table 1.1). The main drivers of interest in the present study that can be reconstructed using palaeo-proxies include fire, herbivory and climate.

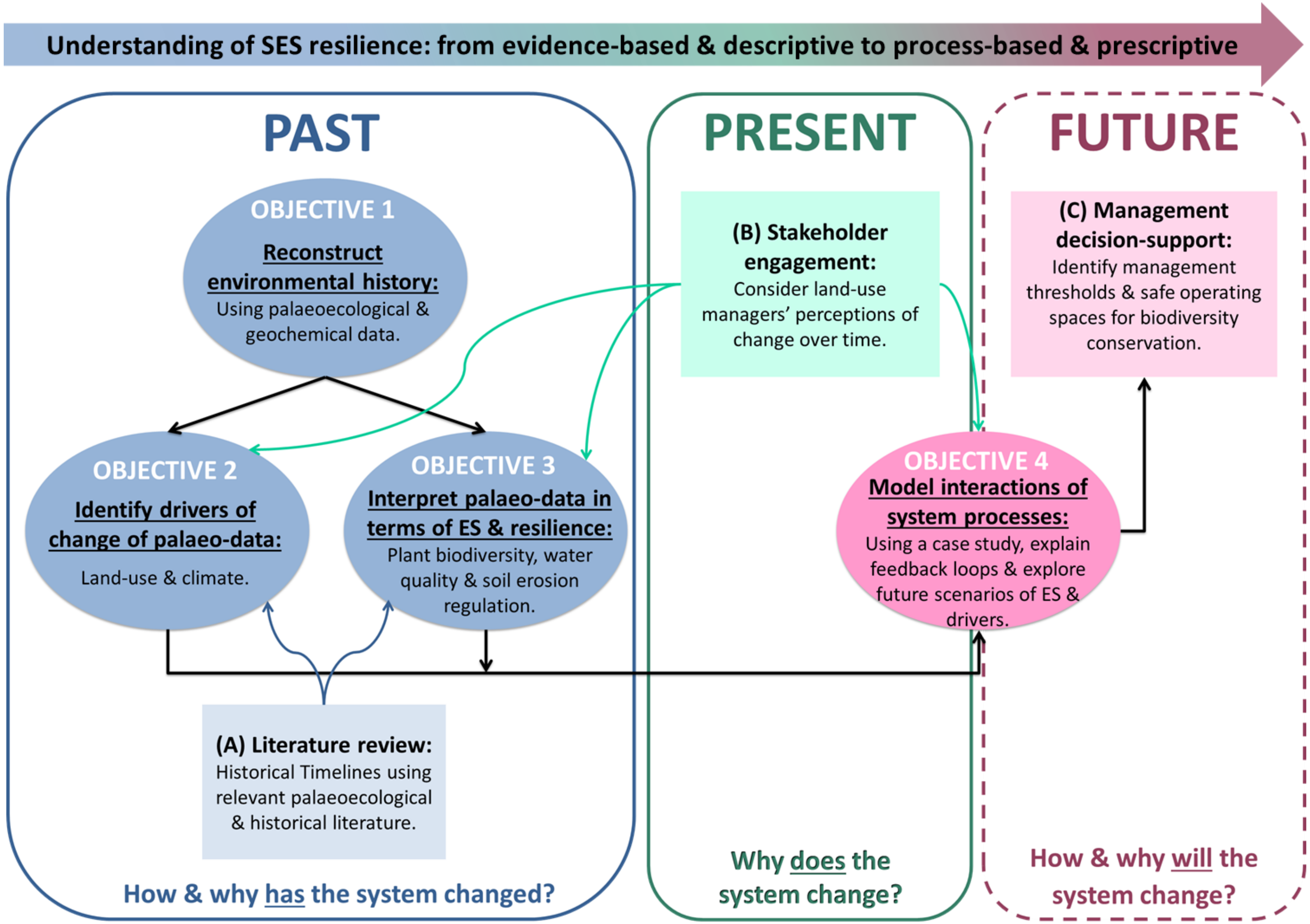



*Figure 1.2: A qualitative and quantitative mixed methods approach to using the past-present-future lens of environmental change as a conceptual meta-framework to explore the concept of long-term changes in ecosystem services and drivers. This illustration outlines the four objectives and three components of this interdisciplinary SES research that can inform ecosystem-based approaches in sustainable land-use management.*

***Table 1.1: Research questions and associated hypotheses (italicized text) to apply the conceptual meta-framework, the past-present-future lens of environmental change. Objectives 1-4 and Component A-B are aligned to factors of complex SESs, signified by a-g, adapted from Dearing et al. (2012, 2015). Association with chapters are also shown with a detailed breakdown per chapter in section 1.5 below. Note that Component A (Chapter 2 and 3) and Component C (Chapter 7) are supporting components needed to explore the rationale of the past and future.***

| | Factors of complex SESs (Dearing et al. 2012, 2015) | PhD Objective/Component | Research Question (RQ) | Hypotheses | Associated chapter in thesis |
|---|---|---|---|---|---|
| PAST | **(a) Reference conditions: Baselines and Past analogues.** | **Objective 1:** To reconstruct environmental history over centennial-millennial timescales by using palaeoecological, physical and geochemical techniques. | **RQ1.** Has plant biodiversity, water quality, soil regulation, fire, herbivory and climate changed over decadal to millennial timescales? | ***H1.*** *Human impact has decreased plant biodiversity, water quality and soil regulation, and increased fire, herbivory since land-use by the early European settlers and continuing into agricultural intensification.* | Chapter 1 and Chapter 2; Palaeo data Chapter 5 and Palaeo data Chapter 6 |
| | **(b) Interactions: Feedback mechanisms** | **Objective 2 and Component A:** To identify drivers of land cover change and interpret the findings in the context of known climate and land-use history (fire, herbivory, crop cultivation and conservation – Component A) in the uplands and lowlands. | **RQ2.** What anthropogenic and environmental factors have influenced/driven these changes in land cover over time, and how have the drivers interacted? | ***H2.*** *Anthropogenic fire and grazing have had increasing influence on land cover in the recent past compared with pre-colonial settlement. Past climate change during the Medieval Climate Anomaly and Little Ice Age will have little impact on land cover and fire regimes compared to 20$^{th}$ C warming, which will have a larger negative impact.* | |
| | **(c) Trends** | **Objective 3:** Interpret palaeo data in terms of ecosystem services (ES) such as plant biodiversity, water quality and soil erosion regulation and resilience. | **RQ3.** What do the results suggest about the resilience and sustainability of ES use at different levels of land-use intensity? | ***H3.*** *ES provision will have been eroded by increasing anthropogenic impact and these changes will be more marked at the lowland agricultural site than at the upland conservation site.* | |
| PRESENT | **(d) Safe operating spaces, and (b) Interactions: Network functioning** | **Component B:** Consider stakeholder perceptions of past and future change, often influenced by policy, governance, and practice. | **RQ-B.** How do stakeholders perceive change in landscapes and interactions between ES and drivers of change over time? | ***H-B****. Multiple land-use managers with local contextual knowledge have a general understanding of ES and environmental change, with a limited long-term palaeoecological and systems thinking perspective.* | Chapter 5 and 6, SD modelling Chapter 7 |
| FUTURE | **(e) Modelling: Dynamic Behaviour, (f) Fast and slow processes, and (a) Reference conditions: Baselines (g) Complex behaviour: Thresholds and regime shifts, and Early warning signals** | **Objective 4 and Component C:** Model future scenarios of changes in select and appropriate ES (plant biodiversity) under various land-use and climate conditions and interpret in terms of future management and conservation - Component C. | **RQ4.1.** Can a system dynamics model simulate the changes in ES and drivers identified in the palaeoecological record? | ***H4.1.*** *If model inputs are set at the historical values, the outputs will match the palaeoecological trends.* | Chapter 7 |
| | | | **RQ4.2.** What are the interactions between system processes, how might these change in the future and what does this mean for sustainable land-use management? | ***H4.2****. Feedbacks between ES, herbivory, fire and climate will have reduced resilience and ecosystem services provision over time.* | |

Previous research identified ecosystem services that are important for people within the CFR (Table 1.2). These ecosystem services are linked to human wellbeing values such as water security, environmental resilience to natural disasters, and respect and care for the natural environment that supports biodiversity conservation, recreation, education, tourism and agriculture (Bengtsson, 2015; Fischer and Eastwood, 2016; Holden et al., 2021; Midgley et al., 2014; Olen and Audouin, 2007). In highly productive landscapes, such as within the CFR, there may be tension between prioritisation and trade-offs of multiple ecosystem services, such as regulating (water quality and soil regulation), provisioning and supporting services (plant biodiversity) competing with other ecosystem services such as food and fibre production that supports people's livelihoods and the economy (Table 1.2). Furthermore, given the interconnectedness within complex SESs, ecosystem processes or states such as biodiversity, water and soil are interdependent, such that if one is threatened then all are at risk of degradation (Bennett et al., 2009; Kremen, 2005; Kremen and Ostfeld, 2005).

***Table 1.2: Examples of Ecosystem Services (ES) provided to the people of the Cape Winelands District of the CFR (adapted from Midgley et al., 2014; Olen and Audouin, 2007).***

| ES process/state | ES/benefits derived from the ecosystem process/state |
|---|---|
| **Biodiversity** | Pollination of crops and natural vegetation; provision of useful species for flower harvesting, wood for fuel, food and medicines; cycling and movement of nutrients, soil stability and soil carbon storage; control of potential agricultural pests; climate stabilisation and moderation of weather extremes and associated impacts; and the provision of aesthetic beauty and recreational opportunities. |
| **Water** | Purification of water and attenuation of floods by wetlands; water supply for drinking, irrigation, and manufacture of products; and the cultural services provision, by rivers and freshwater bodies, of places of recreational, aesthetic, spiritual or religious value. |
| **Soil (and land)** | Provision of nutrients, water and physical rooting support for agricultural crops and natural vegetation, and ecosystem functioning (e.g., a medium for completion of insect life cycles); hydrological functioning such as infiltration of precipitation, runoff control and recharge of groundwater; and the provision of construction and road building material in the form of sand and laterite gravel that are sourced from the soil profile. |

There is potential to include palaeoecological insights in ecosystem assessments and the understanding of resilience and variability of ecosystem services within the CFR context. Therefore, the next stage in the research process is to interpret the palaeoecological data in terms of changes in ecosystem services (Objective 3, Figure 1.2 and Table 1.1 and see section 4.2.1 in Methodological Approach Chapter 4). The present study aims to address this gap in the literature by describing how and why ecosystem services have changed over time in order to increase the relevance and accessibility of palaeoecological reconstructions to relevant stakeholders and research. The main ecosystem services of interest in the present study that can be reconstructed using palaeoecological proxies include plant biodiversity, water quality and soil erosion regulation.

Furthermore, resilience theory is used as a theoretical framing for interpreting changes in ecosystem services and to discuss whether critical ecological thresholds have been crossed. Resilience is defined as the ability to withstand disturbance without loss of critical functions and/or the ability to recover from disturbance (Carpenter et al., 2001; Chapin et al., 2010; Folke, 2006; Folke et al., 2004; Holling, 1973; Lew et al., 2016; Marchese et al., 2018; Quinlan et al., 2016; Walker and Meyers, 2004). Applying resilience theory to palaeoecological data provides an important interface with neoecology and helps ground potential future scenarios in the realm of what is ecologically possible. However, the presentation of palaeo-data is often inaccessible to non-specialists and there needs to be translation of palaeoecological data into useable formats, to demonstrate its significance to stakeholders and to strategically position applied palaeoecology within relevant management, governance and policy frameworks.

### 1.4.2 Stakeholder engagement and systems dynamics to provide insights on the *present* (Component B and Objective 4)

Current threats by climate change and unsustainable land-use practices can cause degradation of ecosystem function, putting sustainable development and livelihoods at risk. Therefore, it is essential to present long-term information that interfaces with current concerns over ecosystem degradation, restoration and management questions in a participatory way. The motivation to qualitatively and quantitatively develop a SD model that represents a SES problem and captures non-linear dynamics and processes is fuelled by the need to move beyond the typical descriptive nature of presenting palaeoecological findings. Therefore, the present study used semi-structured interviews (a key activity for relationship building between the stakeholders and the researcher) and a multi-stakeholder engagement workshop with key informants also present to provide a unique opportunity for participants to network and discuss the interaction of ecosystem services and drivers and how these combinations of feedbacks impact sustainable land-use management (see section 4.2). The social learning techniques provided an opportunity to support local land-use managers in being agents of change in their own work context. Capturing multiple stakeholder narratives of change over time and using this in combination with the palaeoecological reconstructions can elucidate the feedbacks between ecosystem services and drivers for the construction of systems maps and causal diagrams (Component B and Objective 4, Figure 1.2 and Table 1.1). This knowledge was then used in the next research process steps as guidance for integration in a model that simulated interactions between landscape processes.

### 1.4.3 Using system dynamics modelling to provide insights into the *future* (Objective 4 and Component C)

Palaeo-records of ecosystem responses to drivers provide insight into resilience of ecosystems under different disturbance regimes (Folke et al., 2004; Gunderson and Holling, 2002; Holling, 2001, 1973). If palaeo-records of ecosystem services responses to drivers over time suggest that resilience has not been exceeded and critical ecological thresholds have not been crossed then this can help guide management that keeps local drivers of disturbance such as fire and herbivory within their historical range of variability, thereby helping to maintain SES resilience (Carpenter et al., 2001; Chapin et al., 2010; Folke, 2006; Lew et al., 2016; Marchese et al., 2018; Quinlan et al., 2016).

However, as the past is an imperfect analogue for the future, a potential way to test management and policy recommendations for an ecosystem that is currently in, or is approaching, an undesirable state is to use Participatory System Dynamics (PSD) as a method that generates process-based insights and an analysis of future scenarios (Objective 4, Figure 1.2 and Table 1.1). PSD processes reveal the system structure and helps stakeholders identify opportunities to change the system structure (section 2.3 and 2.4). The modelled results can provide management decision-support for land-use managers as it considers long-term changes including defining the historical range of variability which could be defined as “palaeo-safe operating spaces” and then informing possible management thresholds for future planning (Component C, Figure 1.2 and see insights in sections 5.6.2 and 6.6.2 and proof of concept in Chapter 7). Management thresholds such as Thresholds of Potential Concern (TPCs) provide monitoring endpoints of acceptable change, the upper and lower limits of which trigger management interventions that keep the system within a safe operating space. They are set conservatively with the aims of avoiding transitions across critical ecological thresholds (Biggs et al., 2011; Biggs and Rogers, 2003; Gillson and Duffin, 2007; Rogers, 2003).

In order to improve the development and outcomes of new approaches, we can build on previous methods and frameworks that use proxies for ecosystem services and time-series data (Dearing et al., 2015, 2012; Holden, 2017) and that combine time-series data and system dynamics approaches (e.g. Hossain et al., 2017 and Armstrong McKay et al., 2019) to expand our understanding of complex SES (Cooper et al., 2020; Hossain et al., 2020, 2016). Such approaches can provide decision-support for sustainable land-use management by providing insights into how feedbacks between drivers influence non-linear responses in complex systems. The motivation of the present study is to apply the conceptual meta-framework (past-present-future lens on environmental change, Figure 1.2) and build on research that combines palaeoecology and SD in a way that moves beyond research objectives and contributes towards addressing SES problems in the CFR. An example of this is the recent study by Armstrong McKay et al. (2019) that combined soil regulation and soil erosion data obtained from limnological records together with regional historical datasets (e.g. GDP, farm income, crop yield, livestock units) and subsequently used a simple SD model of the English agroecosystem to recreate basic trends of several ecosystem services between AD 1980-2013. Using future scenarios analysis, their SD model investigated the extent to which sustainable intensification can be achieved to ensure food and environmental security in the United Kingdom (UK). It is rare for a SD simulation model to incorporate palaeo-data and provide an analysis of future changes in ecosystem services and drivers (e.g. Armstrong McKay et al., 2019). Therefore, the present study is an uncommon methodology in SES research (Biggs et al., 2021; Clements et al., 2021) as it combines palaeoecology and PSD with the potential to yield significant insights for land-use management in the CFR. Given the scope of this research project, it is not feasible to apply all research objectives (Table 1.1) to all three case studies; this could be a task for future research. Therefore, Elandsberg PNR was identified as an appropriate case study site to explore the PSD modelling aspect of the study. It is fit for purpose as it has a desired temporal and spatial context within the Middle Berg River Catchment. As a lowland conservation site, Elandsberg PNR has an intermediate level of anthropogenic impact that has provided interesting palaeo-ecological insights over a period of centuries to millennia (Forbes, 2014; Forbes et al., 2018). These characteristics are ideal for providing a proof of concept for blending palaeoecology and PSD to advance the state of knowledge of applied palaeoecology.

## 1.5 Thesis Structure

Chapter 1 provides an overview, background and rationale for the research problem, defines the overall aim, objectives, components and briefly explains the methodological approach. The objectives and research process adopted in this study are presented in a conceptual meta-framework in Figure 1.2 and Table 1.1. It also describes how each chapter in this thesis addresses the four objectives of the research process.

Chapter 2 provides a review of the literature that showcases the conceptual (systems thinking and complex-based SESs) and theoretical (resilience, thresholds, regime shift, disturbance dynamics) framing for understanding the present patch-mosaic landscapes in the CRF. This chapter also provides an overview of the palaeoecological literature of the CFR and reviews literature which uses the past, including palaeo-proxies, to understand changes in ecosystem services, and to identify potential reference conditions for the determination of ecological and management thresholds and targets. Chapter 2 also reviews literature of PSD methodology and the application thereof in land and water management globally and locally. The chapter concludes with an overview of how the blending of palaeoecology and PSD approaches can generate new knowledge suitable for ‘boundary crossing’ between disciplines and/or theory and practice and improve decision-support for land-use managers.

Chapter 3 shows how this SES has a long history of climatic, environmental and socio-ecological fluctuations that can lead to river manipulations and negative effects on the catchment area within the multifunctional CFR. The sections provide a contextual setting for the three study sites chosen within an altitudinal and land-use gradient in the study focal area, the Middle Berg River Catchment.

Chapter 4 summarises the methods, which include palaeoenvironmental techniques and participatory system dynamics (PSD) used to investigate the research questions and achieve the overall aim of the research project. A suite of palaeoenvironmental techniques is described and matched to indicators of ecosystem services. The iterative, PSD modelling process is also described in Chapter 4.

Chapter 5 and Chapter 6 describe the results of the vegetation surveys, sediment descriptions and physical proprieties, chronologies and palaeoenvironmental analyses of the upland and lowland sites, GWWA and Rhenostervlei Farm, respectively. These chapters discuss the changes in plant biodiversity/vegetation, water quality, soil erosion regulation, fire and herbivory with regard to drivers of change caused by climate and land-use disturbance. Results are interpreted in relation to changes in ecosystem services and resilience theory before the implications for conservation, agricultural management, and restoration are explored.

Chapter 7 presents a proof of concept for blending palaeoecology and PSD insights. Here the changes of a supporting/provisioning ecosystem service (plant biodiversity) and two land-use drivers of change (fire and herbivory) were investigated by using palaeo-data such as fossil pollen, coprophilous fungal spores and charcoal to define the historical range of variability. A PSD approach including stakeholder engagement was used to unravel the temporal complexity by identifying feedbacks in the dynamic SES structure and exploring future scenarios in response to policy/management strategies and climate change challenges. Following a thorough validation, the model was applied to test the outcomes of different combinations of grazing and burning, highlighting its potential as a management decision-support tool.

Chapter 8 draws on the social theories of boundary crossing and social learning to reflect on the iterative process of developing the decision-support tool for use in participatory scenario planning and sustainable land-use management in the CFR. As the decision-support tool only becomes useful when applied, further focus on packaging the end-products that would be useful to the stakeholders was emphasised. It highlights the implications of key insights and maps out a way forward for the potential development of new SD model prototypes that are enhanced by the palaeo- and expert stakeholder input of this mediated and iterative SES research process to improve the knowledge base. Research limitations and suggestions for future research are an important part of this chapter.

# 2 CHAPTER 2: LITERATURE REVIEW

In this chapter, literature will be reviewed in relation to the proposed conceptual meta-framework as established in Chapter 1, the past-present-future lens of environmental change. This conceptual meta-framework will be used to summarise knowledge and tools that enhance our understanding of changes in ecosystem services and resilience of complex social-ecological systems (SESs) over time. First, conceptual and theoretical frameworks for understanding landscape patterns and system states will be described, specifically complexity thinking, and resilience theory, Hierarchical Patch Dynamics, and non-linearity and feedbacks are discussed. Then palaeoecological records from within the Cape Floristic Region (CFR) will be used to describe the effects of multiple interacting drivers, specifically climate, disturbance and land-use on biodiversity and ecosystem services. Mental models and the Iceberg Model of a multi-level systems perspective are also reviewed. Building on these ideas, I will then briefly review the various international and national system dynamics (SD) applications to illustrate how SD is used to explore the dynamic behaviour of complex social-ecological systems (SESs). In this section, I will also review the benefit of stakeholder narratives of change and of combining stakeholder perspectives with SD modelling. Scenario planning is highlighted as a useful framework to help land-use managers understand the driving forces in the context of the landscapes they manage. The literature review ends by showing how this study combines palaeoecological techniques and participatory system dynamics (PSD), and in so doing, shows its niche contribution to a contextual understanding of long-term change. This understanding informs sustainable land-use management and governance in the multi-functional landscape of the CFR, by focusing on three case study sites within the Middle Berg River Catchment focal area.

## 2.1 Conceptual framing of social-ecological contexts

### 2.1.1 Systems thinking and complexity-based SESs

The Earth's structures and processes are dynamic. We do not live in a steady-state world where change is an exception, but rather a world that includes complexity and uncertainty (Folke, 2006). Together with natural drivers such as climate variability, sedimentation and erosion, fire frequency and intensity and herbivory, there are a host of interacting anthropogenic drivers which affect the environment. These include increased human population, socio-economic influences related to industrialisation, and changes in land-use activities (e.g. agricultural intensification and biodiversity conservation). Complexity has increased with the interconnectedness and interdependencies that we see in industries, economies and nations and organisations of the 21st century. A SES, as a type of complex adaptive system, is an emergent concept that acknowledges the strong connections and feedbacks within and between social and ecological components that determine their overall dynamics (Biggs et al., 2021; Cilliers et al., 2013; Dearing et al., 2011; Game et al., 2014; Preiser et al., 2021; Rogers et al., 2013). Decisions to address real-world issues related to conservation and sustainable development are rarely simple chains of cause and effect (Seymour et al., 2020). Instead, decisions can be viewed as embedded networks influenced by social and environmental factors. It is the purpose of systems thinking to consider the ‘whole’ of a system and relationships within the system and not only the individual parts of the patterns we see over time. Therefore, combining mixed quantitative and qualitative methods to investigate how SESs are interconnected and dynamic within a past-present-future continuum is justified by an overarching conceptual framework such as systems thinking.

Systems thinking uses knowledge, tools and techniques to understand change and unravel complexity by making patterns clearer. Systems thinking can help us identify ways to change patterns effectively to create lasting interventions for chronic problems (Maani and Cavana, 2007) such as ecological degradation and anthropogenic-induced climate change. Therefore, systems thinking allows us to contemplate hidden complexity, ambiguity and mental models (see detailed description in section 2.3). These concepts therefore highlight the need for a systems approach and motivate for inter- and transdisciplinary approaches to research how changes in climate and land-use patterns impact on ecosystem services and SES resilience (Carpenter et al., 2001; Chapin et al., 2010; Fedele et al., 2020; Folke, 2006; Folke et al., 2004; Gunderson and Holling, 2002; Holling, 2001, 1973; Leach et al., 2018; Lew et al., 2016; Marchese et al., 2018; Quinlan et al., 2016; Reyers et al., 2018), and then to apply this understanding to manage real world issues such as biodiversity conservation and sustainable agriculture in the present.

### 2.1.2 Theoretical framing to understand environmental change

#### *2.1.2.1 Resilience theory, thresholds and regime shifts*

Using techniques and tools to understand the hidden complexity of SESs can provide a sense of how resilient a SES is over space and time. The resilience theory perspective emerged from a stream of ecology called ecosystem dynamics, where human actions became a central part of understanding the capacity of ecosystems to generate natural resources and ecosystem services. Resilience was initially defined as the ability of a system to recover from disturbance (Holling, 1973). The following definition of SES resilience with three distinct components is useful in the present study: (1) the spectrum of shocks or amount of disturbance a system can absorb and still remain within a domain of attraction (alternative stable state or regime); (2) the capacity for learning and adaptation, and (3) the degree to which the system is capable of self-organising (Carpenter et al., 2001). Thus, SES resilience is the capacity of a SES to develop and sustain its fundamental function, structure, identity and feedbacks that support a combination of ecological, social, and economic well-being (Chapin et al., 2010; Lew et al., 2016; Marchese et al., 2018). The resilience perspective is also core to systems thinking because it emphasizes non-linear dynamics, thresholds, uncertainty and surprise. It also shows how periods of gradual change interplay with periods of rapid change and how such dynamics interact across temporal and spatial scales (Folke, 2006).

The link between resilience and sustainability is important in the context of the present study. There are many definitions of sustainability, often sectorally-specific, and in the present study, sustainability is defined as system longevity and the ability for environmental resources to meet present societal needs without compromising the ability to meet the needs of future generations (Chapin et al., 2010). The more adaptable and the longer a desired system can be maintained, the more resilient and sustainable it is as it will persist despite changing environmental conditions. Therefore, sustainability of a SES within the changing earth system (e.g. increases in the impact of land-use and climate change) will depend on its resilience (Quinlan et al., 2016). A resilient system can adapt when disturbed and grow in the face of uncertainty and unforeseen disruptions such as is anticipated in the CFR as a result of climate change. Disturbance can cause an ecosystem to undergo a regime shift from one stable state to another after crossing an ecological threshold (Biggs et al., 2018; Scheffer et al., 2001; Seddon et al., 2014). The shift occurs when internal processes of the system (e.g. mortality, growth, decomposition) change and the state of the system (including state variables such as vegetation, fire, herbivory) shifts in a different direction (Walker and Meyers, 2004). A regime shift can either be sudden and dramatic or it can be gradual. Even when a regime shift is gradual (Ludwig et al., 2009) once the threshold is crossed the dynamics of the system shift from one stable state to another as system conditions change rapidly with disproportionally low levels of forcing (Scheffer et al., 2001). Furthermore, some regime shifts may show evidence for nonlinear-reversible fold bifurcations with major time-lags or hysteresis, (Dearing et al., 2015; González Sagrario et al., 2020; Jackson and Wood, 2018). This is an important factor for land-use managers to contemplate for future resilience and sustainable development planning.

If SES resilience and sustainable land-use management is our collective vision, then research needs to consider decision-support for land-use management that concerns defining management thresholds including safe operating spaces, or limits of acceptable change or Thresholds of Potential Concern (TPCs), that prevent undesirable transitions (see section 2.5.2 below). By using the variables analysed, it is valuable to discover the existence of thresholds and what causes them, especially if they have not yet been encountered. By using resilience theory one can explore whether the palaeo-record shows evidence of long-term stability in ecosystem services. Also, despite variations in drivers it is also interesting to determine if a system could reorganise itself if a large disturbance radically alters the structure and function of the system, as explained in the adaptive cycle whereby systems undergo periods of growth, conservation, release and reorganisation (Gillson et al., 2021; Gunderson and Holling, 2002; Walker et al., 2002). Depending on how ecosystem services are managed there is a risk of losing ecological integrity including structure and functionality of a system, which is important for land-use management and future social-ecological resilience. Based on empirical data from high temporal resolution, multi-proxy palaeo-records, what indicators might managers consider when setting management thresholds and monitoring for ecological transitions to undesirable states?

In the case of Fynbos vegetation, palaeoecological data has shown remarkable resilience despite climatic variability and changes in the intensity of land-use, especially at the between-biomes scale and in the uplands (see section 2.2.2 below) (MacPherson et al., 2019; Meadows et al., 2010; Quick et al., 2011). In relating this to resilience theory, Gillson et al., (2020) and MacPherson et al., (2019, 2018) report case studies from the Fynbos-Forest and Fynbos-Succulent Karoo ecotones that highlight the importance of internal reorganisation of plant communities in buffering the effects of environmental change. In these cases, shifts in dominance of various plant functional types help to prevent a critical ecological threshold being crossed. In these examples, the system remained in a particular stability domain, representing a quasi-stable state (Figure 2.1). As aridity increased, for example, instead of a critical transition to a Succulent Karoo dominated vegetation type, Fynbos showed a greater dominance of drought-resistant large Proteoid shrubs. Moreover, in wetter conditions, critical transitions to a Forest dominated vegetation type did not occur. Instead, feedbacks between fire and Fynbos vegetation maintained the Fynbos taxa as dominant elements in the landscape. Further, the vast species richness of Fynbos reinforces ecological character and structural resilience to environmental change through functional redundancy. These mechanisms were invoked to explain why hypothesised critical transitions between biome types did not occur and to set up new hypotheses regarding resilience based on functional redundancy and quasi-stable states between sub-types of Fynbos as opposed to alternative stable states between biomes (Gillson et al., 2020).

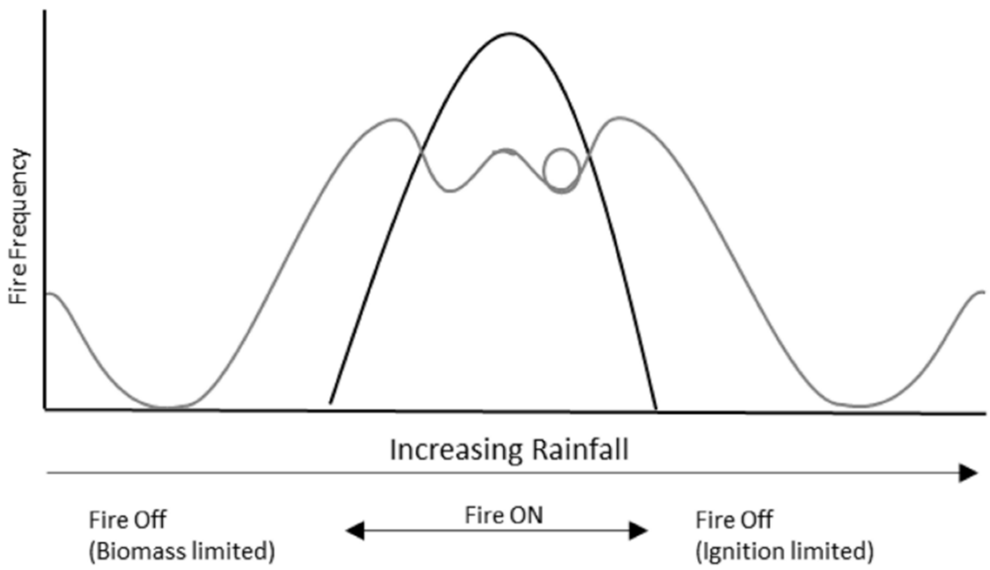



***Figure 2.1: Conceptual diagram of a hypothesised nonlinear relationship between aridity and fire frequency resilience of Fynbos (Gillson et al., 2020; p.9). An example of the ball (current system state) and cup heuristic used in resilience theory where the magnitude of change that a system can absorb without undergoing a regime shift to an alternative stable state. Internal re-organisation within fire-adapted Fynbos leads to the emergence of quasi-stable states (that are more and less drought adapted) and contributes to Fynbos resilience.***

#### *2.1.2.2 Dynamics in patch-mosaic landscapes: disturbance and competition*

A further aspect of resilience in the CFR is the maintenance of patches of distinct vegetation types within a landscape mosaic. For example, Thicket and Forest vegetation represent vegetation communities that are distinct from other vegetation elements such as Fynbos or Renosterveld. A landscape is a heterogeneous area consisting of distinctive patches and these patches are organised into a mosaic-like pattern (Peng et al., 2012; Wu and Loucks, 1995). Patchiness is the result of distinct landscape processes and changes in vegetation elements at spatial and temporal scales, for example, through interactions between fire, topography and local hydrology. The relationship between landscape patterns at a spatial scale and ecological processes over a series of temporal and spatial scales is important for determining the consequences of disturbance (Pauchard and Shea, 2006). Landscape processes include climate change, fire frequency and intensity, sedimentation and erosion, and herbivory, and these ecosystem processes can impact on the size, number, composition, and position of patches in a landscape (Molles, 2015). Adopting a landscape scale perspective is useful in

Fynbos vegetation dynamics as it considers spatial heterogeneity across multiple spatial scales. An example of this is that fires occur at both patch and landscape scale within Fynbos and Renosterveld landscapes. The spatial patterns that emerge as a result of fire history influence herbivory since herbivores will select the most desirable vegetation (Kraaij et al., 2018; Kraaij and Wilgen, 2014; Radloff et al., 2014). Therefore, grazing dynamics could reinforce the impact that fire has on patches and contributes to the overall heterogeneity of the landscape.

As described by Hierarchical Patch Dynamics (Hanski, 1991; Molles, 2015; Walton, 2006; Wu and Loucks, 1995) patches undergo different stages of succession at various ecological scales such that meta-stability of multiple patches promotes resilience and persistence in patch-mosaic landscapes. Therefore, maintenance of all patches (mosaic-like vegetation assemblages or landscape elements) over time is essential to maintaining ecological resilience. Should landscape elements be lost from a landscape, for example, due to excessive disturbance, it may be interpreted that a threshold has been crossed and the resilience of the system has been exceeded. In some cases, however, this can also lead to an increase in the undesired resilience of a system as it becomes more “resistant” (Lake, 2013; Virah-Sawmy et al., 2016) to further climate or land-use change. The intermediate disturbance hypothesis can also be explored when interpreting long-term vegetation change that may arise in fire-prone patch-mosaic landscapes of the CFR. The intermediate disturbance hypothesis suggests the highest biodiversity of consumer-resource ecosystems occurs at intermediate-scale disturbances (Connell, 1975). In community ecology, the competitive exclusion principle states that at most *n* species can coexist on *n* resources such that no more species can coexist indefinitely than the limiting resources (Liao et al., 2007; Loladze et al., 2004). However, the competitive exclusion principle (also known in SD terminology as the “success to the successful” systems archetype) is rarely observed in natural ecosystems. This is because a range of mechanisms exist to maintain coexistence. These include predation and interference between consumers, heterogeneity and disturbance. Therefore, unprecedented rates of disturbance may allow competitive exclusion to reduce diversity due to biophysical constraints such as space, resulting in mechanistic interactions and competition for resources (Connell, 1975; Wang and Liu, 2020; Weiner et al., 2019).

Interacting components at different hierarchical levels such as the feedbacks between fire and herbivory make landscapes ecologically complex. Additional anthropogenic and climate threats add to this complexity. An example of this in Renosterveld vegetation where plant diversity is highest at intermediate frequencies or spatial scales of fire and herbivory (Milton, 2007). Unprecedented large-scale over-grazing or wildfires could disrupt the mosaic-patch structure by competitive exclusion and homogenise landscapes, leading to a loss of ecological character resilience or to the increased resilience of a less desirable state (Forbes et al., 2018). With a greater human impact being evident in the last 150 years, the multi-proxy analyses conducted in the present study may reveal evidence of threshold behaviour, especially in the more disturbed lowland agricultural site at Rhenostervlei Farm (RV3 sedimentary core).

## 2.2 Using the Past: A long-term palaeo-perspective on SES change

This section of the literature review will explore the role of palaeoecological techniques in assessing environmental change, ecosystem dynamics and services. The purpose is to: (1) Highlight the importance of a long-term palaeoecological perspective. (2) The palaeoecological literature review considers past palaeoclimates and vegetation change in southern Africa and in particular the winter rainfall zone (WRZ) of the south-western Cape. The review is embedded into a high level integrative framework (Dearing et al., 2015, 2012) to show the feasibility of using multi-proxy palaeoecological techniques to better understand complex SES and inform in land-use management. (3) Show how applied palaeoecology can evaluate ecosystem services for conservation and land and water management; and (4) Provide merit for applied palaeoecological data as a historical assessment method that captures dynamics over time and provides reference conditions for setting realistic management targets.

### 2.2.1 Benefits of integrating neoecological and palaeoecological perspectives to enhance resilience thinking

In order to make realistic evaluations and plans for sustainable land-use management, it is key to have a comprehensive understanding of the historical range of variability and natural ecological character of ecosystems (Davidson, 2016; Finlayson et al., 2005; Gell et al., 2013, 2018) as well as the effects that land-use practices and climate change have on ecosystem services. It can therefore be argued that applying a past-present-future continuum is crucial in sustainable natural resource management (Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014). This approach can be achieved through combining palaeoecological data with neoecological data to understand the link between pattern and process in present-day landscapes. Further, system dynamics (SD) modelling (including future scenario techniques) can be used to simulate these processes and ultimately provide lessons from the past for ensuring a more sustainable future (Armstrong McKay et al., 2019; Gillson, 2015; Hossain et al., 2016).

Neoecological studies are important for nature conservation and many ecological field data collection techniques are used successfully to provide an ecological context in SES research to understand present day patterns of plant biodiversity (e.g. point counts and quadrats) (Clements et al., 2021). Plant biodiversity is monitored within formal protected areas of the CFR but this is rarely the case in areas that are not formally protected (e.g. on private lands located mostly in the lowland regions). Neoecology shows how existing species interact with their current and recent environment yet lacks adequate temporal scale to reflect longer term-processes (<60 years). Furthermore, monitoring only began in recent decades when substantial alteration of ecosystems had already taken place, therefore, it is necessary to gain a more complete understanding of the effects of climate change and past land use on ecosystems (e.g. Gell et al., 2018). Palaeoecological data are a valuable yet often underutilised temporal source since the recorded timescales precede monitored or surveyed data. These archives provide an improved understanding of ecological patterns and processes. There is a need for a palaeoecological perspective to strengthen neoecological studies and ecological theory, and applied palaeoecology has been recognised as a relevant tool for advancing the understanding of complex SES by capturing an historical account of change over time (Biggs et al., 2021; Clements et al., 2021).

Furthermore, there is potential to combine insights from long-term palaeoecological data with ecological theory, providing an interface between neoecology and palaeoecology. Specifically an understanding of the resilience and variability of ecosystem services can be enhanced by the combination of neoecology, palaeoecology and resilience theory (Gillson, 2015). Palaeoecological methods complement neoecological methods in developing ecological theory that adds value to management strategies based on resilience and thresholds. Neoecological data can also verify and therefore strengthen findings from the palaeoecological record. The use of experimental plots with different fire and herbivory treatments to assess vegetation dynamics is a neoecological approach to test conceptual models of environmental change and resilience that may be revealed in the palaeoecological data (Forbes et al., 2018).

Studies show that short-term perspectives, often based on collective societal memory and reflections, can provide an unrealistic interpretation of environmental baselines and change (Gillson, 2015; Wolfe et al., 2012). Neoecological studies are characterised by a short-term perspective which considers an annual to decadal timescale of usually less than 60 years. However, many ecological processes take place on longer timescales than this and some SESs with long human histories may exhibit responses to human impacts over millennia (Dearing et al., 2012). Therefore, a long-term palaeo-perspective that considers decadal, centennial and millennial temporal scales is needed. A longer perspective provides improved evidence of the historical range of variability and an opportunity to identify drivers of environmental change such as land-use disturbance as a result of urban development, agriculture and conservation and/or climate change (Birks, 2012; Davidson, 2016; Dawson, 2011; Finlayson et al., 2005; Gell et al., 2013, 2018; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014). A vital part of knowledge brokering for research into practice is to communicate insights derived from the palaeo-records and modelled results that may be controversial. As an example, Dearing et al. (2012) found that agricultural intensification was the main multidecadal driver of losses in regulating services on a regional scale in the lower Yangtze Basin. However, improved environmental policies and regulation after the late AD 1980s helped stabilize losses of biodiversity and erosion regulating services after AD 1990. As interdisciplinarity is recommended as a potentially effective way to apply long-term palaeoecological perspectives, it is inevitable that innovative approaches and methodologies will emerge, as evident in the agricultural lands of the Lower Yangtze basin in China (Dearing et al., 2012; Zhang et al., 2015) and in the United Kingdom (Armstrong McKay et al., 2019).

### 2.2.2 Long-term records to understand complex SESs in the CFR during the Holocene

Several archaeological studies contribute to our knowledge of past human impact within the CFR (Deacon, 1992; Smith, 1987) but multi-proxy palaeoecological studies are limited to a few sites scattered within the CFR (Figure 2.2). These noteworthy sites include the Central Cederberg (Sneeuberg Vlei and Driehoek Vlei) (Meadows et al., 2010; Meadows and Sugden, 1993, 1991a), Elands Bay Cave (Cowling et al., 1999; Parkington et al., 2000), De Rif (Quick et al., 2011), Klaarfontein Springs (Meadows and Baxter, 2001), Pakhuis Pass Shelter (Scott and Woodborne, 2007a, 2007b), Princess Vlei (Neumann et al., 2011), Verlorenvlei (Stager et al., 2012), Elandsberg PNR (Forbes et al., 2018), Groenkloof (MacPherson et al., 2018), and Platbos 1 (MacPherson et al., 2019).This section highlights findings of a few of these reconstructions, which provide some understanding of vegetation change and palaeoclimates within the winter rainfall zone (WRZ) of the CFR. Their geographical location is shown in relation to the three case study sites in the present study, Groot Winterhoek Wilderness Area (GWWA), Rhenostervlei Farm and Elandsberg Private Nature Reserve (Elandsberg PNR) (Figure 2.2). Findings are summarised in Table 5.1 and Table 6.1, which is used later in the thesis to assist with the reconstruction of environmental history and interpretation of ecosystem service responses to drivers in this study.

Furthermore, the reconstructions are integrated into an adaptation of an integrative framework which includes factors (labelled a-g) related to the dynamic behaviour of complexity-based SESs (Dearing et al., 2015, 2012). See Table 1.1 for how these factors are considered in relation to the thesis structure: (a) Reference conditions: Baselines and Past analogues, (b) Interactions: Feedback mechanisms and Network functioning, (c) Trends, (d) Safe operating spaces, (e) Modelling: Dynamic Behaviour, (f) Fast and slow processes, and (g) Complex behaviour: Thresholds and regime shifts and Early warning signals. This review approach allows us to view environmental change in the CFR as the product of interactions between different system variables over the last ca. 10 000 years and therefore the opportunity to assess dynamic behaviour, through comparisons of trends, interactions between drivers and complex behaviour including non-linearity and thresholds. The framework proposed by Dearing et al., (2015, 2012) will also be used to summarise the findings and recommendations generated from the present study's conceptual meta-framework in an integrative manner that is useful for informing management - see Synthesis and Conclusions Chapter 8, Table 8.1).

#### *2.2.2.1 Fast and slow processes/rates and trends of environmental changes from the Last glacial inter-glacial transition (LGIT)*

The transition from LGM and LGIT to early-Holocene was a time of rapid environmental change. Even though the study sites in this thesis do not cover the LGM, the transition to early-Holocene provides an opportunity to assess the responsiveness of Fynbos vegetation to climate change and interaction with other processes. A recent review investigated the rates of vegetation change over the past 18 000 years using a global palaeo-dataset (> 1100 fossil pollen records), and found that increased rates of change are a general global phenomenon in plant biodiversity. The rates of change accelerated strikingly during the Late Holocene (approximately 4 600-2 900 thousand years ago) (Mottl et al., 2021).

The fossil pollen record from Pakhuis Pass Shelter (Figure 2.2) analysed by Scott and Woodborne (2007b) at shows that increased ambient temperatures and the low soil moisture during the LGM supported low Asteraceous Fynbos shrubs (*Stoebe/Elytropappus*-type pollen) and "pure fynbos" (Ericaceae, Proteaceae, *Passerina* and *Cliffortia* taxa). The transition of vegetation from the LGM to the early-Holocene showed a shift to Asteraceae, *Dodonaea* and *Olea* with Aizoaceae-type succulents, which suggest a Fynbos-Thicket mosaic. Sites like Pakhuis Pass Shelter in the lower-lying rain shadow of the Cederberg is more susceptible to climatic changes and therefore variability in the pollen record. As the Central Cederberg and De Rif sites are closer to the upland conservation site, GWWA (Figure 2.2), they may serve as a suitable comparison for the Holocene environmental reconstruction from the present study .In contrast to the state shift to Fynbos-Thicket mosaic shown in the Pakhuis Pass Shelter record (Woodborne, 2007b), pollen data as a proxy for vegetation and stable isotope data as a proxy for climate from De Rif (Quick et al., 2011) show that during climatic transitions from the LGM to early-Holocene (ca. 19 500-11 500 14C yr BP) mountain Fynbos communities remained relatively stable. Quick et al. (2011) reported that the difference between the stable pollen assemblage as a vegetation change proxy and the variability in the stable isotope records as a climate proxy obtained from De Rif is explained by mountain Fynbos being strongly influenced by geological constraints of oligotrophic sandstone substrates. Thus, the stable isotope signatures represent changes in precipitation and/or fire regime that did not affect the overall Fynbos community composition. This is further shown by the construction of rainfall envelopes for mountain Fynbos taxa in comparison with non-Fynbos taxa which illustrated the resilience of mountain Fynbos in response to climate change (Quick et al., 2011). These slower changes in vegetation and climatic conditions are also confirmed by other environmental reconstructions of the Central Cederberg during ca. 9500 to 1300 cal yr BP (Meadows et al., 2010; Meadows and Sugden, 1993, 1991a).

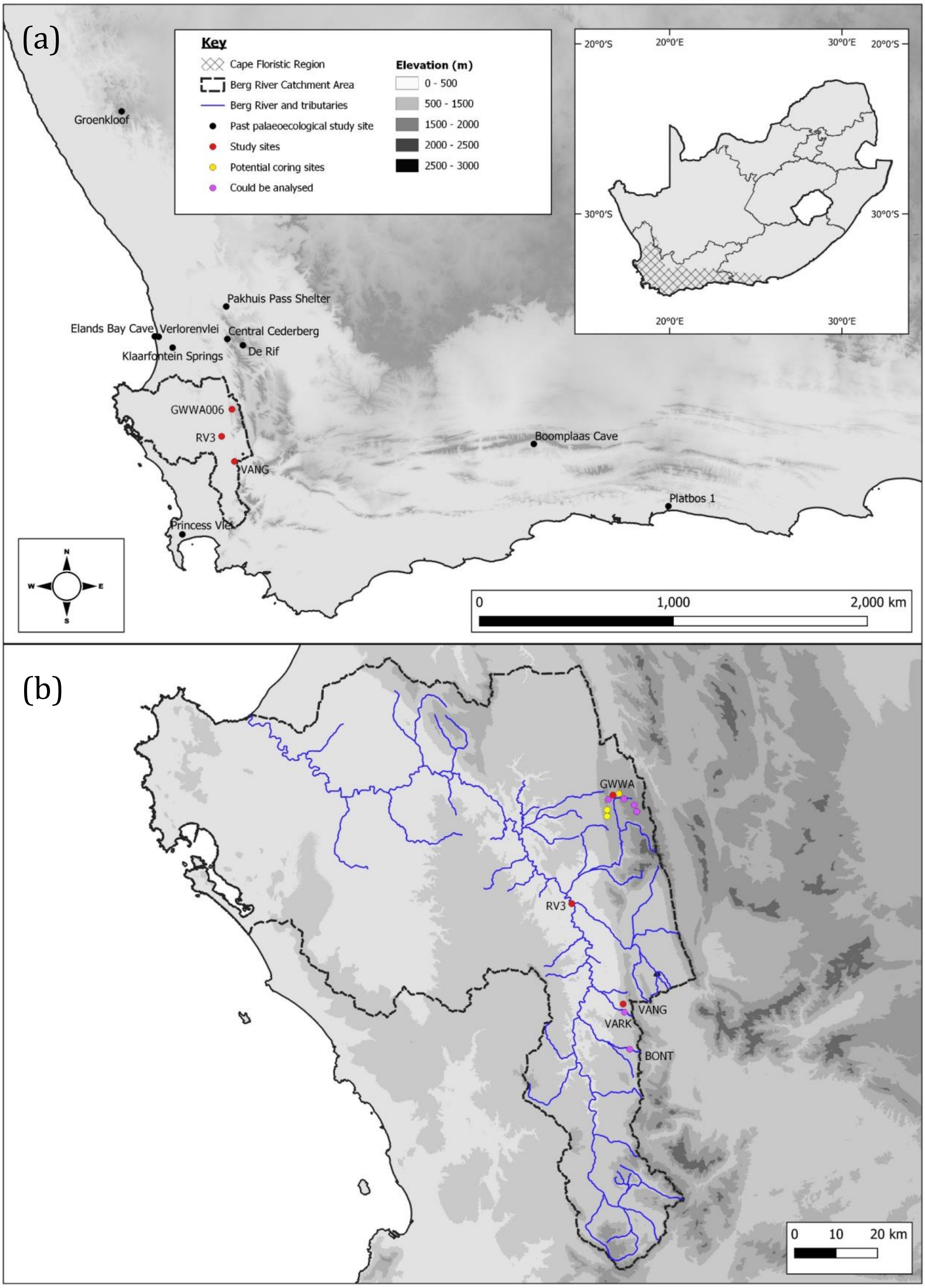



*Figure 2.2: Map showing (a) the Berg River Catchment Area in relation to the three case study sites and other noteworthy palaeoecological study sites mentioned in the literature. (b) Map showing wetlands within the understudied Berg River Catchment that have already been cored and could be analysed using palaeoecological techniques (pink dots – see section 3.4) and potential sites that could be cored in future research (yellow dots).*

The 4150-year palaeo-record from Princess Vlei (Figure 2.2) in the Cape Flats region improved the vegetation reconstruction of the late-Holocene by analysing a high temporal resolution record (average 1 sample/70-year) of Fynbos vegetation and climatic (Neumann et al., 2011). The Princess Vlei palaeo-record indicated a dry climate in ca. 4150-3400 BP suggested by the abundance of Asteraceae, *Crassula* and Aizoaceae pollen and decreased aquatics and sedge pollen. Increased relative abundance in *Morella*, Cyperaceae and *Carpacoce* pollen in ca. 3400-2600 BP suggests a more humid climate. Thereafter there was a shift back to drier environments from ca. 2600-1900 BP followed by wetter conditions in ca. 1850 BP. Interestingly, no Renosterveld vegetation elements were captured in the Princess Vlei pollen record despite being situated in the Cape lowlands. This further highlights the importance of understanding the changes in ecosystem dynamics at the two lowland sites, Rhenostervlei Farm and Elandsberg PNR, in the present study.

The results suggest that fynbos vegetation persisted throughout the warming/drying of the early Holocene. However, the previous resilience of the mountain fynbos to climate change does not necessarily rule out instability under future climate change scenarios (Gillson et al., 2020; MacPherson et al., 2019; Quick et al., 2011). Moreover, Quick et al. (2011) suggest that the contrasting palynological records found at the central Cederberg (Meadows et al., 2010; Meadows and Sugden, 1993, 1991a) and the northern Cederberg site at Pakhuis Pass Shelter (Scott and Woodborne, 2007b, 2007a) is because of low temporal and/or taxonomic resolution of pollen data. Therefore, the pollen records cannot detect plant species turnover related environmental parameters, which may disguise changes in vegetation communities (Chase and Meadows, 2007; Meadows and Sugden, 1991a). However, these studies show a general trend of vegetation change according to climate variability within the CFR, and show ecological resilience achieved via the internal reorganisation of Fynbos vegetation (Gillson et al., 2020).

The present study will add to the limited palaeoecological record within the Middle Berg River Catchment of the CFR. Lowland Fynbos communities near coastal or biome boundaries may be more sensitive to future climate change (Roberts et al., 2001). One could argue that the palaeo-record within the lowlands would reveal that landscape processes are faster in the lowlands than the upland mountainous regions. Thus, the reconstruction of environmental history from GWWA and Rhenostervlei Farm will contribute to our understanding of the resilience, or otherwise, of this vulnerable, globally-recognised biodiversity hotspot in light of future climate change scenarios.

#### *2.2.2.2 Reference conditions/states linked to climate resilience: the Mid-Holocene Altithermal (MHA), Medieval Climate Anomaly (MCA) and Little Ice Age (LIA)*

Palaeoenvironmental records illustrating the interactions between climate, fire and increasing land use intensity during the Holocene are recognised as useful guides for the sustainable management of resilient landscapes and ecosystems as they suggest that environmental change has been the norm (Roberts et al., 2001). Determining which parts of the multi-functional landscape of the CFR have historically been sensitive or resilient in the face of biophysical drivers such as climate change and anthropogenic drivers can assist current governance and management determine early warning signals to prevent ecological thresholds being crossed that are detrimental for SES resilience. Specifically, the interplay between known climatic variations throughout the Holocene and increasing anthropogenic influence can be investigated using multi-proxy approaches (Bennion et al., 2011a; Dearing et al., 2012, 2011; Jackson and Hobbs, 2009). Such findings are site-specific given the variation in rainfall, temperature, aspect, geology at the local spatial scale, therefore, it is essential to keep this in mind when data handling of pollen records (see section 3.3 and 4.1.3.2 later).

The Mid-Holocene Altithermal (MHA), also known as the Mid-Holocene Climatic Optimum was a postglacial interval centred ca. 5500 years ago. In the CFR, the MHA experienced temperatures that were warmer than at present (Meadows and Baxter, 1999) and is reportedly linked to changes in the mean latitudinal position of the austral westerlies as a response to changes in the extent of sea-ice during the winter months in Antarctica (Chase et al., 2015; Chevalier and Chase, 2015; Meadows and Baxter, 1999; Neumann et al., 2011; Roberts et al., 2001). Furthermore, extreme climatic fluctuations in the last two millennia include the warm and arid Medieval Climate Anomaly (MCA) which occurred from ca. AD 900-1400 and the subsequent colder and wetter Little Ice Age (LIA) from ca. AD 1400-1800 (Lee-Thorp et al., 2001; Mayewski et al., 2004; Nicholson et al., 2013; Stager et al., 2012). Known climatic variations including the MHA, MCA and the LIA, followed by 20$^{th}$ Century warming, have interacted with changing fire regimes and increasing land-use intensity, starting approximately 2000 BP due to the presence of Khoikhoi pastoralists. Historical sources such as ships' logbooks and diaries of visitors to the Cape describe the Khoikhoi people that used to live along the western and southern Cape coasts with livestock such as sheep and cattle. The archaeological and anthropological literature hypothesised that the southward movement of this immigrant population with their livestock was influenced by the environment, the need to exploit new pastures for livestock and cultural and agricultural activities such as acquiring stock from their neighbours (Cooke, 1965; Deacon, 1992; Elphick, 1977; Humphreys et al., 1998; Westphal, 1963) (see section 3.1.2). Such archaeological and anthropological evidence is supported by the palaeoecological record at Klaarfontein Springs where

Poaceae pollen decreased while micro-charcoal increased since 1900 BP (Meadows and Baxter, 2001) (Figure 2.2a). However, when European Settlers arrived in the Cape in the mid-17th century they had an even more significant impact on land-use for centuries (Hoffman, 1997). European Settlers expanded northwards beyond the Berg River and further eastwards towards the Hottentots Holland Mountains by AD 1700 and occupied the region south of the Orange River by the end of the 18th century (Bergh and Visagie, 1985). By the early 19th century, European Settlers were utilising valuable natural resources mostly, mineral and agricultural, which may have had an effect on salinity, nutrients and sedimentation rates, associated with rivers and wetlands in the Berg River catchment.

Although the MHA is an imperfect analogue for contemporary climate change as the drivers and resources use are completely different, it would probably provide a reasonably useful past analogue for a future warmer and drier Western Cape experienced at present. Aridity associated with a poleward drift of the prevailing westerly winds and anthropogenic-induced climate change most likely caused this warmer climate (Toggweiler and Russell, 2008). Moreover, reference to states and processes during warmer and drier conditions of the MCA can also offer appropriate reference conditions. Stager et al. (2012) presented a high-resolution, decade-scale regional winter rainfall zone (WRZ) record over the last 1400 years through the analysis of lacustrine diatom records from Lake Verlorenvlei (Figure 2.2). The diatom record interpreted by Stager et al. (2012) confirmed that the MCA was warm and arid and inferred a wet LIA for the WRZ. There was high precipitation in ca. 1400-1200 BP (ca. AD 550-750), which later decreased until ca. 950 BP (ca. AD 1000). During the LIA there were precipitation maxima in ca. 600, 530, 470, 330, 200, 90, and 20 BP (ca. AD 1350, 1420, 1480, 1620, 1750, 1860 and 1930) (Stager et al., 2012). This analysis is relevant to simulations of modern climates and may help in the prediction of future climate scenarios.

Many CFR conservation-orientated publications use the term 'pristine' when describing remnants of uncultivated land. Using ‘pristine condition’ as a concept in biodiversity conservation contexts can be extremely problematic since the term can be subjective and depends on a certain temporal frame of reference – i.e. shifting baselines (Forbes et al., 2018; Gillson, 2015; Manzano et al., 2019; Pauly, 1995; Wolfe et al., 2012). As an example, the ‘pristine’ composition of the highly endangered Renosterveld is uncertain. The lack of environmental history and appropriate reference conditions or states make it difficult to build evidence-based and process-based arguments for conservation management and restoration. As a result, for example, there is uncertainty about whether ‘pristine’ Renosterveld contained a higher proportion of $C_4$ grasses compared to shrubs and whether Renosterveld today could be considered a “degraded Grassland” (Cowling et al., 1986; Thom, 1952) or whether it was comprised of a combination of shrubs and $C_3$ and $C_4$ grasses as indicated by isotope data from Overberg Renosterveld in the east of the CFR (Curtis, 2013; Curtis and Bond, 2013). Furthermore, the multi-proxy palaeoecological study of Renosterveld at Elandsberg PNR indicated that relative grass abundance was generally lower than expected (Forbes, 2014). Therefore, researchers and practitioners are encouraged to differentiate between the concept of a ‘pristine condition’ and that of a ‘reference condition’ (Bennion et al., 2011b). Additional studies, such as the present study, both within- and between-biome scale throughout the Fynbos biome can contribute to the debate on the historical range of variability and to the determination of suitable reference conditions in disturbance-driven landscapes of the CFR. Such insights are likely site-specific given the variation in rainfall, temperature, aspect, and geology at the local spatial scale, and will also influence the management of vegetation types.

In summary, it is inaccurate to state that all remaining fragments within the CFR should be restored to a ‘pristine’ condition without an improved understanding of how some impact (e.g. pre-colonial), minimal impact (i.e. pre-pastoral) or no impact (pre-hunter-gatherer) on vegetation composition and ecosystem function varied. Instead, each restoration site requires an analysis of long-term environmental change to ascertain the natural historical range of variability (see section 6.6 and section 7.4) and it is important that each management and restoration target is suitable at multiple scales. Therefore, known local and regional historical records of land-use and climate can be used to compliment palaeoenvironmental records. Knowledge of the CFR’s Holocene variability, especially since the mid-Holocene, provides reference conditions that can be used to inform appropriate management and restoration targets for land and water management. The compilation of historical timelines can be examined to substantiate interpretations of the palaeoecological records, as intended in the present study for from GWWA and Rhenostervlei Farm, Table 5.1 and Table 6.1) (Component A, Figure 1.2). The concept of reference conditions/states is discussed in more below as it pertains to identifying and setting management targets that protect ecosystem services for sustainable land-use management.

### 2.2.3 Reference conditions/states and setting realistic management targets

Jackson et al. (2009) shrewdly suggest that all management and policy decisions are based on prevailing societal values, both economic and aesthetic. As an example, a 'historical aesthetic' conservation or restoration target would be one where the ecosystem is in its ‘natural/pristine’ state with natural ecological character (Gell, 2017) . However, unprecedented climate conditions and novel stressors such as increased human population and industrialisation may cause desired conditions to be unattainable management targets (Jackson and Hobbs, 2009). Therefore, decision-makers need to be aware of what is ecologically possible and set targets accordingly, keeping in mind future resilience in the long-term. Additionally, restoring ecosystems to targets identified as reference conditions in the palaeoecological record could be expensive and may even result in the system becoming less resilient or ill-suited to cope with future change (Millar et al., 2007). Thus, the power of applied palaeoecology lies in its unique position to provide a long-term perspective which is often missing from management strategies as well as providing present managers with a variety of conservation or restoration goals.

Gell et al. (2005) conducted a study in wetlands across southeast Australia using diatoms in lake sediments. Their findings suggested that these wetlands are characterised by decreased salinity where previously estuarine areas had been cut off from the sea, as well as increases in pH, turbidity, nutrients and sedimentation rate due to the active regulation of river wetlands that occurred after European settlement (Gell et al., 2005). One of the strengths of the study is that it demonstrates the usefulness of mixed assemblages of allochthonous and autochthonous fossil diatoms (Dixit et al., 1992) to infer fresh water supply and regulation that are ecosystem services related to processes/states of connectivity and turbidity over decadal-millennial timeframes. However, a limitation of this methodology is the use of the fossil diatoms record to infer lotic change that accumulates in a lentic system. In other words, this study cautions against drawing conclusions regarding fast renewal in water supply such as a river in a system which naturally experiences the trapping of water in a ground depression such as a wetland as an example of a lentic system (Gell et al., 2005). Although both types of systems have a fresh water source, weathering and climatic conditions can lead to a wide range of salinities in these two different systems and this needs to be considered when drawing conclusions from the palaeo-data. In the present study, this consideration is especially relevant for the lowland agricultural site which occurs in a depression adjacent to the Berg River (see section 3.2.2 below). The most common interpretation of the dominance of aerophilic species in fossil records is that they are representative of shallow waters and possibly drought events. However, peaks in both aerophilic species and sedimentation rates may be a reflection of flooding events, resulting in increased erosion and surface runoff (Gaiser and Rühland, 2010; Gell et al., 2007; Van der Putten et al., 2008).

If a desirable historical reference condition is identified in the palaeoecological record (section 2.2.2.2), for example, a reference point with the same climate as today or lower levels of disturbance, it could be a modern analogue and an example of a suitable conservation management or restoration target (Bennion et al., 2011b; Dearing et al., 2015). In other words, managers could develop management thresholds that guide the manipulation of current levels of land-use activities such as fire and herbivory to match the levels in the reference condition, thereby ensuring that no critical ecological thresholds are crossed. Since the past is an imperfect analogue for the future, however, an adaptive approach is essential so that managers can adjust management interventions and thresholds as new conditions arise (Gillson and Marchant, 2014).

Reference conditions identified in the palaeoecological record can be prioritized based on what is realistic and feasible, both socially and ecologically. Such long-term data helps ground potential future scenarios in the realm of what is ecologically possible. Multiple stakeholders such as scientists, land-use managers and policymakers can discuss the scientific findings in a tangible way. As an example, stakeholders are more willing to consider multidecadal droughts, landscape-altering fires, and rapid species invasions as realistic possibilities for the future since palaeoecological evidence shows that they have happened in the past (Jackson et al., 2009).

Despite the effectiveness of using palaeoecological studies to investigate reference conditions and establish appropriate restoration targets, Bennion et al. (2011) also caution against attributing variations in the palaeoecological record to anthropogenic activity such as increased grazing or fire due to land-use change as opposed to natural variability. If there have been changes, such as climate change, which have taken place between the reference condition and the present day, this will limit the use of the reference condition as a restoration target (Bennion et al., 2011a). However, it could be contended that if there is known historical data for a study site that can provide some indication of the anthropogenic activities such as historical records of crop cultivation, fire history, herbivore management, and conservation practices, then the chances of attributing variations incorrectly could be minimised. Moreover, the breadth of multiple palaeoecological proxies (Dearing et al., 2012; Jeffers et al., 2015) have allowed researchers to cross boundaries between neoecological and palaeoecological research and therefore show the potential to use multiple palaeoecological datasets to obtain a holistic understanding of environmental change.

### 2.2.4 Using palaeoecological techniques to assess ecosystem services and drivers to inform land and water management

The general context of this study is based on changes in three of four categories of ecosystem services recognised by the Millennium Ecosystem Assessment (2005): supporting/provisioning services, plant biodiversity, and regulating services, including water quality and soil erosion regulation, which are often less well monitored. The drivers are fire, herbivory and climate change (see Table 4.4). Ecosystem services are the essential attributes of an ecological system upon which all humankind depends (Gillson, 2015; Kremen, 2005) and are categorised by the Millennium Ecosystem Assessment into four categories: supporting, provisioning, regulating and cultural (Millennium Ecosystem Assessment, 2005). The importance of palaeoecological research for conservation is well established (Birks, 2012; Dearing, 2008; Dearing et al., 2012; Gil-Romera et al., 2010; Gillson and Willis, 2004; Willis and Birks, 2006). However, few studies evaluate the continuity of ecosystem services over time and how raw palaeoecological data should be translated into relevant 'currencies' required for ecosystem services assessments so as to provide information on the resilience and persistence of ecosystem services (Dearing et al., 2012; Jeffers et al., 2015). For example, Dearing et al. (2012) compiled an extensive palaeoecological proxy index for more than 50 representative ecosystem processes or system states and the index is grouped by the four categories of ecosystem services. Jeffers et al. (2015) further highlights a subset of palaeoecological proxies that can therefore be applied in the tools and modelling approaches used by landscape and ecosystem managers. Palaeoecological, palaeohydrological and palaeoclimate studies have shown the usefulness of certain palaeoecological proxies for water management including water quality, climatic proxies, and the impacts of land-use on erosion and nutrient cycling (Dearing et al., 2012; Gell, 2012; Gell et al., 2013; Jackson et al., 2009; Van der Putten et al., 2008; Wolfe et al., 2012).

While the review paper by Dearing et al. (2012) presents an extensive list of palaeoecological proxies, new techniques of deriving and interpreting proxies regarding the reconstruction of past ecosystem processes or states are continuously being developed. For example, coprophilous fungal spores, which are not included in the Dearing et al. 2012 review, give an indication of herbivory since these spores are associated with fungi that grow on animal dung. Coprophilous fungal spores such as *Sporormiella*, Sordariaceae, *Gelasinospora* and *Coniochaeta* are commonly used to give an indication of changes in herbivory over time (Baker et al., 2013; Carrión et al., 2000; Davis and Shafer, 2006; Gelorini et al., 2011; Graf and Chmura, 2006; van Geel and Aptroot, 2006). Furthermore, Jeffers et al. (2015) notes that as new palaeoecological proxies are developed, it is essential that they are validated against modern ecological measures. Thus, there is a need for more calibration studies particularly in African landscapes (Duffin et al., 2008; Duffin and Bunting, 2008; Julier et al., 2021). Moreover, it is important to note that palaeoecological proxies are not meant as a replacement for monitored or surveyed data, but rather to verify and complement such records and to provide a longer-term estimate of change over time (Dearing et al., 2012; Jeffers et al., 2015) (see summary of palaeo-proxies for ecosystem services used in the present study in Table 4.4). This is substantiated using other data collection methods that explore cultural services such as biocultural diversity and heritage, education values, nature-based recreation, inspiration, knowledge systems, social relations and spiritual/religious values. Therefore, monitored or surveyed data is still valuable for informing sustainable land-use management within the CFR (Holden et al., 2021).

A longer-term perspective (centennial-millennial timescales) can help build more contextual and evidence-based arguments for sustainable land-use management as part of a mixed methods approach that could be helpful to policymakers and land-use managers when protecting ecosystem services and function. Though monetization is commonly used to value ES and explore trade-offs and synergies thereof (Food and Agriculture Organization (FAO) of the United Nations, 2016), quantification of ecosystem function is difficult and the approach presented in this study can complement sustainable decision making by including a past-present-future perspective that is currently lacking from most ecosystem function and services assessments.

## 2.3 Understanding the Present: A multi-level understanding of complex adaptive systems

The past-present-future lens of environmental change as the proposed conceptual meta-framework builds on a foundational concept used by researchers to investigate social-ecological systems (SES) – a “past-present-future continuum” (Birks, 2012; Dawson, 2011; Gillson, 2015; Gillson and Marchant, 2014; Marchant and Lane, 2014). With this foundation in mind, it is possible to consider how current patterns of vegetation distribution or system states reflect long-term processes (e.g. climate and disturbance) as well as the abiotic template (e.g. topography and soils). This understanding of the link between past patterns and processes and present landscape pattern or current system state underpins the exploration of future trajectories and scenarios. We need to have a multi-level understanding of complex adaptive systems. and how natural and anthropogenic drivers interact to influence the ecosystem and the supporting, provisioning, regulating and cultural services it provides. This section of the literature review will provide an overview.

A useful way to understand what a systems perspective entails is to consider the ‘Iceberg Model’ that includes multiple levels, usually ranging from three to six levels. The simplest Iceberg Model includes events, patterns of behaviour, and systemic structures. Monat and Gannon (2015) apply the Iceberg Model to both a natural system and human-designed system by including a fourth level, which contains the ‘physical and chemical forces’ in natural systems and ‘mental models’ in human-designed systems

(Figure 2.3) A Mental Model of a dynamic system is defined as "a relatively enduring and accessible, but limited, internal conceptual representation of an external system (historical, existing or projected) whose structure is analogous to the perceived structure of that system" (Doyle and Ford, 1998, 1999). This Iceberg Model argues that events and patterns that are more likely to be easily observed are caused by systemic structures and mental models or physical and chemical forces, which are often hidden. It represents a broad context which demonstrates how underlying systemic structures impact our daily lives in observable ways. Thus, this perspective goes beyond dynamics as it considers the psychology behind the structure (Monat and Gannon, 2015). The value of having a systems perspective is that it assists in identifying points of leverage for decision-making and intervention. The point of leverage increases as one moves from events to systemic structure (Figure 2.3).

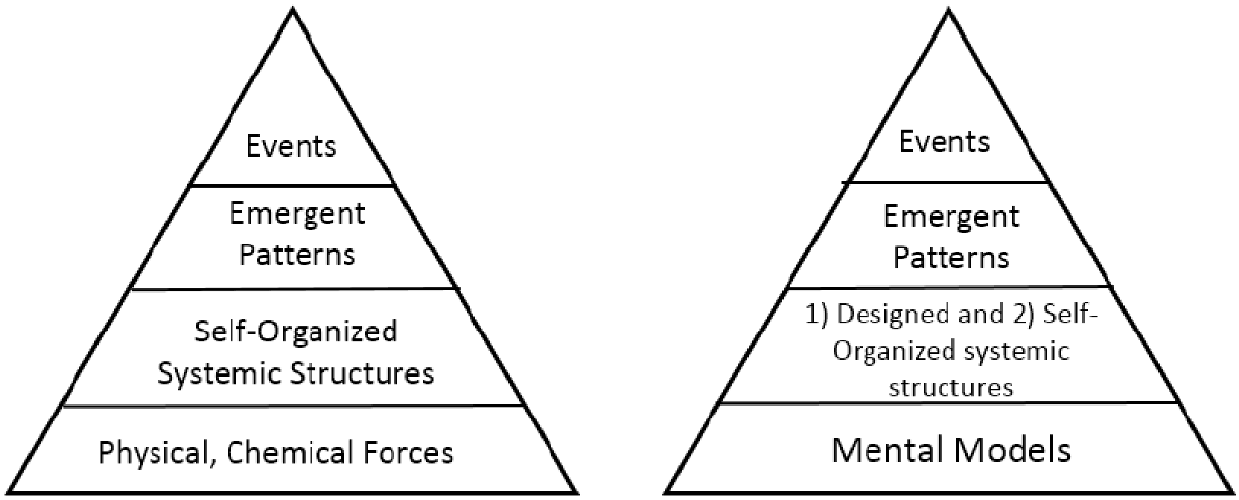



***Figure 2.3: The Iceberg Model which depicts multiple levels of a systems perspective as applied to a natural and human-designed system by Monat and Gannon (2015); p.18.***

The tip of the iceberg depicts Events ('E') or consequences and these are occurrences that can happen on a day-to-day basis, for example, grazing or a veld fire. As events or consequences are at the tip of the iceberg, they are easily observable and are what often drive decision-making. This level of the systems perspective can be described as the reactive elements of the system. Events and decisions are influenced by Patterns ('P') or behaviours which is the next level of the iceberg. Patterns are defined as sets of consistent and recurring observable events that may be physical, behavioural, or mental. Patterns are characterised as being adaptive in nature and are usually a result of the underlying systemic structures and forces (Monat and Gannon, 2015). A biophysical example of patterns would be veld fires occurring mostly during the summer or when human activity is high.

Systemic Structures (‘SS’) are ways in which the parts of a system are organised. This structure generates the patterns and events and decisions observed, and are thus generative in nature. In other words, if one focuses on the systemic structure and design at the system level, as opposed to merely reacting to events, there is more leverage to influence and change the system that shapes our future. It is argued that similar results will occur even if different people are in the same structure. Thus it is the structure that causes 85% of the problems not the people (Monat and Gannon, 2015). Therefore, in order to understand patterns of behaviour, one should first identify and understand the system structures and underlying mental models (human-designed systems) or physical and chemical forces (self-organised natural systems) (Figure 2.3). The following factors need to be considered at the systemic structures level: causal or feedback loops and delays; interrelationships; accumulations or stock-and-flow thinking (e.g. increases in population size or resources); policies that govern decisions; authorities and approval levels; organizational hierarchy; social hierarchy; rules and procedures; goals and metrics; reactions and the incentives and fears that cause them; and underlying forces that exist in an organization or physical environment (Monat and Gannon, 2015). In the case of the present study, leverage points are places to intervene in the system that influence feedback loops of ecosystem services and drivers that cause the SES problems described by stakeholders. Meadows (2008) cautions that because leverage points is are not intuitive, misuse of them can systematically worsen whatever problems one is trying to solve. Furthermore, delays are often sensitive leverage points for policy, if they can be made shorter or longer (Meadows, 2008). Therefore, system delays are an essential aspect to consider when evaluating changes in ecosystem services and developing policy and management strategies for maintaining ecological function and SES resilience.

A Mental Model (‘MM’) gives insight into why the systemic structures exist and how they may be changed (see definition by Doyle and Ford (1999, 1998)). The following factors are considered when investigating mental models: perceptions or beliefs; pressures which stem from perceptions; and affects such as emotions, attitudes, feelings, rationalities and irrationalities. Therefore, the present study will use the interrogation of high resolution, multi-proxy palaeo-data together with insights derived from stakeholder engagement to explore the connections between P, SS and MM and perhaps use insights to identify potential leverage points that could be activated to change management and future patterns. A multi-level systems perspective as illustrated in Figure 2.3 encourages one to establish some ‘distance’ from certain events and decisions. Investigating patterns that emerge over time, which resulted in the events and decisions, therefore, gains a better understanding of the systemic structure and the system dynamics at play.

In summary, the resilience theory perspective evolved when scientists used models as a tool for understanding and for incorporating stakeholders and interest groups in adaptive management and learning of ecosystem processes (Folke, 2006). Adopting resilience theory as a perspective for SES analysis is useful as it includes the understanding of social processes and social learning (Currie, 2018; Reed et al., 2010; Roux et al., 2017) including knowledge–system integration, adaptive capacity and visioning and scenario building, that allow for management of essential ecosystem services. Thus, multiple levels of systems thinking together with resilience theory allows for a broader interpretation of 'SES resilience' and its context which includes integrated system feedbacks and cross-scale dynamic interactions (Folke, 2006). This comprehensive understanding of the past and present can provide a framework for investigating policy and management implications.

## 2.4 Looking into the Future: Applying a systems perspective and participatory system dynamics (PSD) to inform management and restoration

The following section highlights that the main purpose of a systems perspective is to help understand and address real-world problems. In terms of the proposed conceptual meta-framework (Figure 1.2), the systems perspective and particularly participatory system dynamics (PSD) modelling is essential in developing a perspective on future change as a) it builds on the descriptive multi-proxy palaeo-data of environmental change and evidence-based understanding of past change to develop a process-based understanding of change over time; and b) it uses this process-based understanding to simulate possible future scenarios. The review will consider the following: language and methodological tools used in systems thinking and modelling - particularly the systems dynamics methodology; examples of where SD has been applied in natural resource management and land-use planning to influence a system; and how relevant stakeholders and their management systems can benefit from this approach. The examples span the following environmental management interventions: freshwater resource management, fire management, conservation agriculture, fisheries, ecological restoration, and scenario planning.

### 2.4.1 Applications of system dynamics in land and water management

During the late AD 1950s, a type of systems thinking and modelling methodology known as system dynamics (SD) methodology began to emerge. SD falls within a framework that respects and fosters the needs and values of awareness, openness, responsibility and equality of individuals and teams (Maani and Cavana, 2007). Within this framework, the purpose of SD is to solve problems and create more robust systems, which minimise the likelihood of unpleasant surprises that are often inevitable and have unintended negative consequences. The underlying assumptions of SD are that the system is dynamic, consists of on-going interdependencies and closed loop relations between different social-ecological factors, and it is therefore structured and operational (Clifford-Holmes et al., 2017b; Richmond, 1993). The SD methodology uses the theory of information feedback systems and mathematical models to simulate complex systems in order to understand decision-making processes (Forrester, 1961; Sterman, 2000). A system can also be visualised via 'diagramming languages' such as systems maps and causal loop diagrams (see section 2.4.2 below) that depict the feedback structures, for example, interactions between variables such as ecosystem services and drivers of change within the system.

Several international studies demonstrate the strengths of using SD to manage multiple resources and sectors in sustainable land management (Bennich et al., 2018; Brzezina et al., 2017, 2016; Herrera de Leon and Kopainsky, 2019; Menendez et al., 2020; Stave and Kopainsky, 2015; B. L. Turner et al., 2016; Turner and Kodali, 2020) and water management (Simonovic, 2009; Stave, 2003; Tidwell et al., 2004; Wang et al., 2011; Winz et al., 2009). It is important to note that disturbances include environmental, economic, and social shocks or stressors, or deliberate policy interventions intended to improve the SES's capacity to buffer shocks or stressors. Stave and Kopainsky (2015) successfully demonstrate how SD can be used to examine mechanisms and pathways of vulnerability when exposed to various disturbances that affects not only the environment but also food security and population health. Furthermore, they have also shown the usefulness of SD research for sustainable land-use management. As an example, a South African-based SD study by Crookes et al., 2013 showed how SD can be used in portfolio mapping and risk analysis to guide decisions on the feasibility of restoration activities. Their simulation model considered trade-offs between restoration costs and ecological, hydrological and economic benefits associated with restoring ecosystem function (Crookes et al., 2013). Additional noteworthy South African-based studies have shown the usefulness of SD research in both marine management (Weller et al., 2016, 2014) and freshwater management (Carnohan et al., 2021; Clifford-Holmes et al., 2017a, 2017b).

SD insights can help policymakers and land-use managers better appreciate the interconnectedness of the SESs in which their governance authorities reside. This is especially true since socio-economic factors and fire-grazing-vegetation dynamics (Collins et al., 2013) coupled with climate change vulnerabilities may challenge seemingly rational management decisions and policies at multiple scales. Therefore, innovative modelling approaches may assist in gaining insight into future uncertainties and support improvements in resource management of complex, feedback driven ecosystems (Turner and Kodali, 2020). In the case of fire-prone Mediterranean-type ecosystems such as the CFR, a multi-functional landscape that consists of conservation, agricultural and urban land with varying levels of disturbance (see section 3.2), vegetation-fire dynamics have implications in both the environmental and political sectors (Hoffman et al., 2019; van Wilgen et al., 2012). Thus, determining a balanced management approach to fire suppression and prevention, including controlled patch burning, could prevent the negative reinforcing consequences of long-term fire disturbance in this biodiversity hotspot.

The purpose for using SD goes beyond the typical technicalities of developing a simulation model which is often expert-driven or for the purpose of prediction or forecasting (Clifford-Holmes et al., 2017b; Epstein, 2008). Benefits to modelling interactions and system processes include the identification of core dynamics and trade-offs, the discovery of novel questions and the training of practitioners. Clifford-Homes et al., (2017b) present an overview of why SD methods are appropriate for informing integrated and participatory water management governance within an African context. They also provide a unique evaluation of several SD South African-based case studies between AD 1980-2016. Their evaluation was based on a conceptual framework for distinguishing between the diverse motivations for modelling that occur along four continua: (1) analysis to recommendations; (2) understanding; (3) democracy in action interventions; and (4) intervention. Moreover, with the prediction of climate change uncertainty and potential future threats (Haensler et al., 2011; Hoffman et al., 2011; Midgley and Thuiller, 2007), there are meaningful SD applications that inform climate change adaptation and decision-support tools in South Africa (Brent et al., 2017a, 2017b; Carnohan et al., 2021). Therefore, the present study has the potential to add to this sum of knowledge as it aims to use mixed methods to unravel the physical and social complexities that interact over several spatial and temporal scales within the Middle Berg River Catchment (see Figure 1.1, Figure 3.1, Figure 3.2, and Figure 3.3 and section 3.1.3).

### 2.4.2 Benefits of using tools to collectively understand complex SESs into the future

#### *2.4.2.1 Participatory approaches and dynamic- and causal-thinking*

To achieve sustainable ecosystem management, the process by which information is derived and utilised needs to include stakeholder participation and mutual learning (Knight et al., 2008; Wheeler et al., 2019). Managing SESs is increasingly challenging since boundaries in spatial, social and ecological contexts are indistinct. However, participatory approaches can cultivate opportunities to navigate the dynamic complexities as a collective. One of the significant advantages of using SD is the integration of multiple sources of information when developing models (Ford, 2000). These range from ‘hard’ numerical sources obtained from experiments or archives through to so-called ‘soft’ sources from written case studies, expert opinion, stakeholder knowledge and personal intuition. SD provides a unique mathematical framework for integrating both physical and social processes important to integrated catchment management. Therefore, participatory system dynamics (PSD), which combines stakeholder engagement with typical SD methods, is advocated in the present study to increase the value of results regenerated. Studies have shown that PSD can be effective in addressing contentious topics such as water resource management and regional planning issues that often have high potential for conflict (Tidwell et al., 2004; Winz et al., 2009).

The SD methodology, including causal loop diagrams (CLDs) and stock and flow diagrams (SFDs), is one of the systems approaches that benefits from “diagrammatic reasoning” (Hoffmann, 2011). Such an approach reduces vulnerabilities and enhances resilience by assisting stakeholders in clarifying assumptions, structuring a problem space, or identifying unexpected implications of an unplanned disturbance or an intentional policy intervention. Collectively reflecting and co-creating a process-based understanding (as per the multi-stakeholder engagement workshop - see section 4.2.3.1) and using systems maps and CLDs to build theory diagrammatically provides a conceptual model of the SES problems at present and how they may change over time. Together with CLDs, stock and flow diagrams (SFD), can help in building a richer, process-based picture of a system (Bou Nassar et al., 2020; Inam et al., 2015; Langsdale, 2007; Videira et al., 2010; Voinov and Bousquet, 2010).

In reality, many problems are created internally, caused by unintended consequences of our decisions and actions, as well as a result of our mental models (Maani and Cavana, 2007). To truly achieve understanding and leverage in land-use management, stakeholders need to adopt an endogenous point of view as it would have the highest net payoff (Richardson, 2011). As there is an appetite for SES research innovation (Clements et al., 2021; Preiser et al., 2021), sustainable land-use management in the CFR could benefit from insights generated from the proposed mixed methods approach that applies the past-present-future lens on environmental change. However, if SES consultants, researchers or activists would like to pursue PSD approaches, then several best practices and pitfalls should be considered: problem definition and setting the model boundary; stakeholders' participation in defining the project goals; model testing by the end-users; personal and social learning; and revision of models to support adaptive management (Winz et al., 2009). The most pertinent aspects of the systems thinking approach are indisputably causal/feedback loops and stocks and flows, however, endogenous thinking is extremely important when applying the skills and thinking required to understand endogenous sources of complex system behaviour with the underlying assumption that the system is the cause (Anderson and Johnson, 1997; Richardson, 2011). Key to practicing an endogenous point of view is constant examination and transparency around values, habits, beliefs and assumptions and creating a safe space for engaging in creative dialogues regarding both the endogenous and exogenous forces that shape and influence management. Therefore, combining stakeholder engagement with typical SD methods (systems maps, CLDs and SFDs) can assist with the exploring challenges such as high variability in water supply contrasted with SES challenges related to droughts, flooding, agricultural irrigation and urban water demands that exhibit important short-term and long-term effects common to arid/semi-arid environments, such as the Berg River Catchment.

New approaches and processes whereby researchers and stakeholders iteratively define research questions based on real-world problems, use mixed methods, and deliberate on the relevant evidence and end-products is the fundamental nature of sustainability science  (Kates et al., 2001). A wonderful example of this within the applied palaeoecology community is showcased in a series of case studies from the Rocky Mountain region of the western U.S. Here, relevant water resource stakeholders and scientists were able to engage and incorporate palaeoenvironmental insights into water management and policy (Jackson et al., 2009). They found that stakeholders were more willing to consider SES dynamics (e.g. multidecadal droughts, landscape-altering fires, and rapid species invasions) as realistic possibilities for future scenarios since palaeoecological evidence shows that they have happened in the past. Adapting global examples such as this will help researchers tailor approaches and processes that are suited for conservation and natural-resource management in southern Africa.

#### *2.4.2.2 Future scenario analysis to generate management insights*

The opportunity to extend our observations to pre-colonial impact conditions and assess the degree to which the system is presently outside or within the historical range of variability or natural ecological character is valuable for defining more realistic target conditions for management. Examples of a useful long-term palaeoecological perspective includes the use of the MHA or MCA as possibly warmer climate analogues, extending our knowledge back to before the great acceleration and the importance of improved information on pre-colonial reference conditions or baselines and "natural ecological character" (Dearing et al., 2015; Forbes et al., 2018; Gell et al., 2005; Gell, 2012; Gell et al., 2013, 2018; Manzano et al., 2020; Newall et al., 2015). It is increasingly difficult to attribute causes or drivers when there are multiple co-occurring drivers that might result in unprecedented changes. An example of this is the multi-proxy approach by Jackson et al., (2009) used to provide complementary palaeo-evidence on climate change and ecological responses over the last 15 000 years in the Rocky Mountain region of the western U.S.. By using multiple proxies they were able to conclude that human land clearance and fossil fuel combustion dominated climate forcing during the last 150 years, as opposed to no influence from human processes such as natural variability of the carbon cycle in the previous 11 000 years. Therefore, palaeoecological studies that use multiple palaeo-proxies allow for possible decoupling of many drivers and responses. We can develop an evidence-based understanding which is important for developing a process-based understanding which together inform potential future scenarios.

The documentation of changes over time, whether derived from palaeoecological or neoecological data collection, emphasises the complexity of multiple drivers and interactions that epitomize complex SESs. However, the use of statistical bivariate and multivariate analyses (Dearing, 2008; Zhang et al., 2015) as well as modelling tools in addition to time series data could assist with untangling driving and responding variables responsible for complex behaviour. Complex, process-based dynamic models attempt to replicate the intrinsic variability of the natural environment, ecosystem functioning and to predict future change. Bringing together neoecology and palaeoecology aspects can assist in forming the basis for future bioclimatic modelling predictions (Anderson et al., 2006) as well as being the basis for exploring changes in ecosystem services under different scenarios of climate and land-use. Application of a long-term palaeo-perspective in the CFR offers the scope for developing and testing socio-ecological models that can provide alternative future scenarios.

Thus, the analysis of future scenarios that incorporates qualitative and quantitative data obtained from palaeoecology, neoecology and stakeholder engagement (Colles et al., 2009; Louys, 2012; Maani and Cavana, 2007; McCann et al., 2006; Rull, 2010) and assumptions based on theoretical thinking (e.g. Hierarchical Patch Dynamics, theories of complexity and resilience, social learning) can assist in informing policy and management. Therefore, the conceptual meta-framework proposed by the present study, the past-present-future lens of environmental change, will be beneficial for biodiversity conservation and conservation agriculture by supporting planning before implementing large-scale land management strategies (e.g. CapeNature's Protected Area Management Plan (PAMP)).

Integrating ecosystem services priorities into land-use planning and decision-making in non-biodiversity sectors will entail a paradigm shift and require the implementation of various strategies. Potential noteworthy strategies include the co-production of knowledge and building collaborative partnerships for consolidated ecosystem-based management, bridging the gap between data modelling and information for better policy decisions, and communications, outreach and education (UNEP, 2015). Another strategy to emphasise the importance of changes in ecosystem services over time could be via combining future scenario analysis with participatory techniques to evoke multi-stakeholders' ambitions, plans and perceptions of change. Future scenario planning or scenario analysis is an alternative model to the conventional predictions approach that develops precise, quantitative assessments of future environments, which is often hindered by compounded uncertainties and qualifications (Biggs et al., 2008, 2007; Bohensky et al., 2006; Caves et al., 2013; Gray et al., 2018; Palomo et al., 2011; Peterson et al., 2003). Participatory future scenario planning allows for the co-creation of a new understanding of present situations and for stakeholders to build shared visions of possible future developments. The interaction of multiple knowledge systems thus stimulates dialogue on adaptation measures that can help land-use managers adequately plan for the future (Jackson et al., 2009; Phillips et al., 2009; Rice et al., 2009; Wollenberg et al., 2000).

Although future scenario planning is more commonly used in the business and finance community, Walker et al. (2002) propose a four-step participatory approach to investigating SES resilience for regional scale management (Figure 2.4). Thus, recognising that managers are an integral part of the system and need to be included in the modelling process to address complex SES problems and achieve long-term sustainable land-use management. By foregrounding a stakeholder-driven description of the system and issues leads to a set of potential future scenarios that capture the major uncertainties in the system's future dynamics and improved management decision-support. Therefore, futures analysis techniques (e.g. Figure 2.4 and Figure 2.5) help advance our understanding of change and uncertainty in complex SESs because people are encouraged to think constructively and systematically about multiple futures depending on the complex, unpredictable interactions of stakeholders, institutions, ecological processes of the system and its dynamics (Hichert et al., 2021).

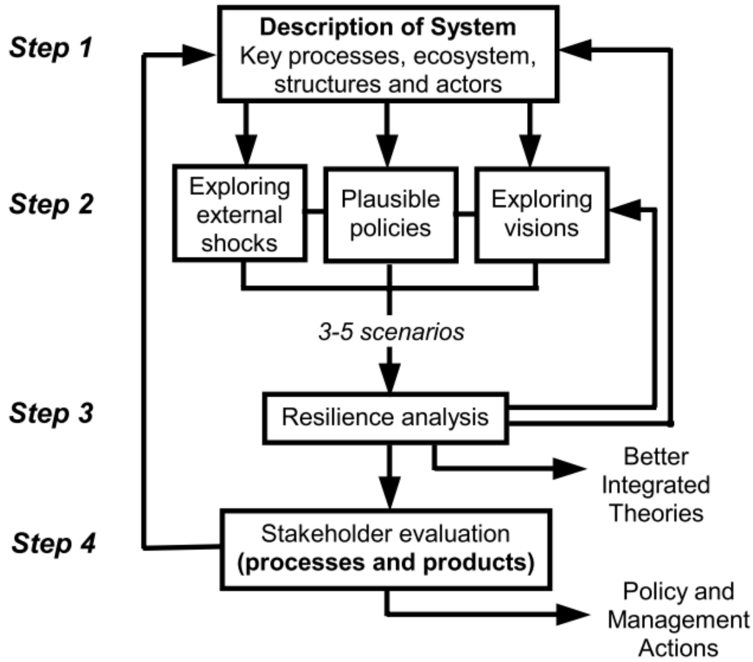



***Figure 2.4: A participatory approach using a four-step framework for the analysis of resilience in social-ecological systems (Walker et al., 2002; p.9).***

Furthermore, scenario planning also allows for the effective use of palaeoecological findings in decision-making (Jackson et al. 2009). Jackson et al. (2009) show how alternative scenarios can be used as a starting point for exploring ecosystem vulnerabilities under a range of future conditions, and as a means for examining how conservation strategies might unfold in situations where there are multiple drivers of change (Figure 2.5). Each quadrant represents an alternative future scenario with a unique combination of climatic and fire regimes (Jackson et al., 2009). Each scenario provides a contrasting story of the impact that these environmental drivers will have on species and ecosystems and therefore provide a launchpad for examining the relative trade-offs (Chisholm, 2010; Dearing et al., 2019; Holden et al., 2021; van Wilgen et al., 2016b)**,** and costs and benefits of various management interventions such as climate change mitigation or adaptation measures. The alternative future scenarios can range from low impact to high impact (Figure 2.5). The highest impact for example may illustrate the ecosystem crossing a threshold and transitioning to an alternative stable state which is analogous to a period in the palaeo-record that occurred 10 000-5000 BP or a ‘novel ecosystem’. An alternative scenario illustrating a slightly lower impact, may show the ecosystem transitioning due to increased frequency of ‘inevitable surprises’ over time.

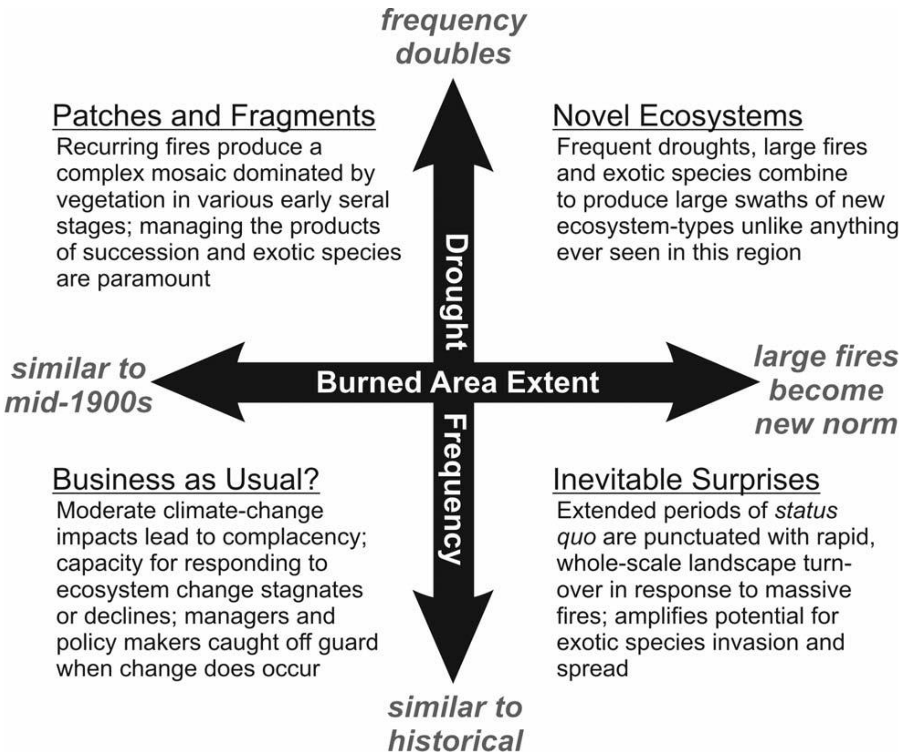



***Figure 2.5: An example of a scenario planning matrix (cited by Jackson et al. 2009; p.75). Each axis represents a critical driver of system change or a significant trend in the environment.***

Owing to the complexity of SES and the uncertainty of future environmental conditions, approaches are needed that allow stakeholders to explore systems dynamics through time and the effects of future changes in environmental or social factors. Well-informed strategic ecosystem-based management plans would benefit from resilience and management frameworks such as those proposed by Walker et al. (2002) and Jackson et al. (2009).

Stakeholder participation is effective in problem scoping and policy analysis within the realm of environmental decision-making processes (Videira et al., 2010). Voinov and Bousquet (2010) highlight several stakeholder-based modelling approaches including, inter alia, participatory modelling, group model building, mediated modelling and collaborative learning. Participatory system dynamics (PSD) used in the present study included a mediated or facilitated approach (van den Belt, 2004; van den Belt and Blake, 2015), which promotes the understanding of cross-level and cross-scale links, the co-creation of salient, credible, and legitimate knowledge that encourages the crossing of boundaries that often hinders learning between multiple sectors or perpetuates the research-implementation gap (Knight et al., 2008). Participation from multi-stakeholders creates an opportunity for social learning as it reinforces the articulation and evolution of shared desirable visions, goals, and community capacity development for the future (Helfgott, 2018; Roux et al., 2017; van den Belt and Blake, 2015; Videira et al., 2010; Voinov and Bousquet, 2010). Owing to the complexity of SES and the uncertainty of future environmental conditions, approaches are needed that allow stakeholders to explore systems dynamics through time and the effects of future changes in environmental or social factors. Well-informed strategic ecosystem-based management plans would benefit from resilience and management frameworks such as those proposed by Walker et al. (2002) and Jackson et al. (2009).

## 2.5 Boundary crossing to advancing the past-present-future lens of environmental change

### 2.5.1 Congruence with the wider sustainability policy context

It is essential for boundary crossing to take place in order to understand complexity-based SES resilience and to improve land-use management decision-support. This requires cross-sectoral and multi-stakeholder initiatives to bridge the research-implementation gap. Therefore, I place my study within the wider sustainability policy context through the concept of ecosystem services, which is well established in the field of conservation biology and sustainability science. South Africa’s National Climate Change Response White Paper (Government of South Africa, 2011) recognises that stressed ecosystems will compromise ecosystem services and thus the ability for SESs to respond to climate change risks. Therefore, the use of ecosystem services as part of an overall strategy can help societies adapt to the negative effects of climate change, whilst resulting in socio-economic and biodiversity/ecological co-benefits that contribute to sustainable development (Aronson et al., 2019; Pasquini and Cowling, 2015; Secretariat of the Convention on Biological Diversity, 2009; Vignola et al., 2009).

Within the United Nations Framework Convention on Climate Change (UNFCCC), the 2015 Paris climate agreement and the United Nations 2030 Sustainable Development Goals (SDGs) highlight the urgent need for scientific progress in understanding and modelling the impact of increased human influence. Strong and cooperative partnerships for change at local-global governance levels is required (e.g. goal number 17 of the 17 SDGs). Therefore, ecosystem-based management and governance is a suitable integrated approach for applying polycentricity by matching governance levels to the scale of the SES problems under investigation (Biggs et al., 2012; Fedele et al., 2020; Folke et al., 2007; Leach et al., 2018). Research shows that both top-down and bottom-up modes of governance (and sometimes a combination of both) have been successful in addressing sustainability goals (Elena M Bennett et al., 2015; ISSC et al., 2016).

Sustainable development cannot be attained if economic development is prioritised and natural resources are mismanaged (e.g. overgrazing, inappropriate fire regimes, overuse of pesticides, ploughing close to buffer zones). As a result, ecological degradation could occur and the ecosystem's integrity would be compromised, leading to unsustainable land-use in the long-term and a loss of ecosystem services (Dearing et al. 2012). By definition sustainable development is a social aspiration as well as an environmental one (Bond and Morrison-Saunders, 2011; Grace, 2019; Morrison-Saunders et al., 2015), therefore, social innovations that aim to mainstream palaeoecology in the sustainability dialogue will add value. Achieving sustainability goals requires intensive, long-term stakeholder engagement that includes investing time and resources into building relationships within civil society and other sectors, knowledge brokering, and supporting local champions (Carcaillet et al., 2001; Schneider et al., 2021). A polycentric governance system can enhance the protection of ecological infrastructure and resilience of ecosystem services provision with multiple, nested governing authorities at different scales (Ostrom, 2005).

### 2.5.2 Using resilience theory to integrate palaeoecology and PSD modelling insights that inform the identification of ecological and management thresholds

Managing complex SESs requires the balance of diverse social and ecological needs for sustainability, and the resilience concept offers a conducive framework for policy recommendations to be articulated in a prescriptive manner (Duit et al., 2010; Herrera, 2018, 2017). However, before we can address the prescriptive usage of resilience in landscape management and governance, taking stock of the state of systems (levels of resilience, stability and degradation) and the processes that underpin them is imperative. Here, considering how stakeholders perceive change in landscapes and interactions between ecosystem services and drivers of change in comparison to the palaeoecological records would be relevant for thinking of 'safe and just operating spaces' (Dearing et al., 2015, 2014) and setting management thresholds in the CFR. Therefore, land-use management could exist well away from critical ecological thresholds.

The key difference between ecological thresholds and management thresholds is that the latter provide safeguards for preventing an ecological threshold from being crossed (Biggs and Rogers, 2003). Therefore, management thresholds precede ecological thresholds which may not be easily reversible if the system changed to an alternate stable state and shows hysteretic characteristics (González Sagrario et al., 2020). If management thresholds are crossed due to unsustainable land-use practices that are detrimental to ecological integrity, management action needs to change to avoid crossing critical ecological thresholds that risk the ecosystem's long-term health and function. Furthermore, calibration of proxy-data with modern analogues together with stakeholder consultation are required to set and operationalise management thresholds. When considering the resilience of an ecosystem, it is important that managers and other stakeholders can envision possible future scenarios where ecological thresholds are and are not crossed. For example, decreased fire and grazing may be required to maintain all landscape patches, whereas overgrazing and an increased fire regime may result in some landscape patches disappearing and the homogenisation of undesirable vegetation in the landscape. A further important consideration is that management thresholds must be set more conservatively than ecological thresholds, i.e. management interventions must precede and act to prevent the transition to an undesirable state. The ecological thresholds and management thresholds are thus distinct, an idea pioneered by South African National Parks in the strategic adaptive management in the Kruger National Park (Rogers, 2003). Furthermore, the distinction between ecological thresholds sand management thresholds is critical in accommodating the preferences of different stakeholder or user groups, and the concept of Thresholds of Potential Concern (TPC) allows different stakeholders to express their preferred ecological state, within what is ecologically feasible (Biggs et al., 2011). TPCs are defined as a management tool that defines limits of acceptable change for upper and lower levels along a continuum of change in selected environmental indicators (Biggs and Rogers, 2003).

Variability is intrinsic to the concept of TPCs. It is assumed that key ecological parameters vary, and the purpose of developing TPCs is to keep these parameters within limits that are deemed ecologically and socially desirable (Biggs et al., 2011; Rogers, 2003). However, long-term variability was not explicitly included in the original formulation of TPCs. Therefore, Gillson and Marchant (2014) (Figure 2.6) proposed integrating long-term data from palaeoecology and other disciplines so that TPCs are based on a realistic understanding of ecological variability and resilience over time. Palaeo-data can therefore potentially be used to explore the ecological character of ecosystems (Davidson, 2016; Finlayson et al., 2005; Gell et al., 2013, 2018), information which can be integrated into the development of TPCs (e.g. Gillson and Duffin, 2007).

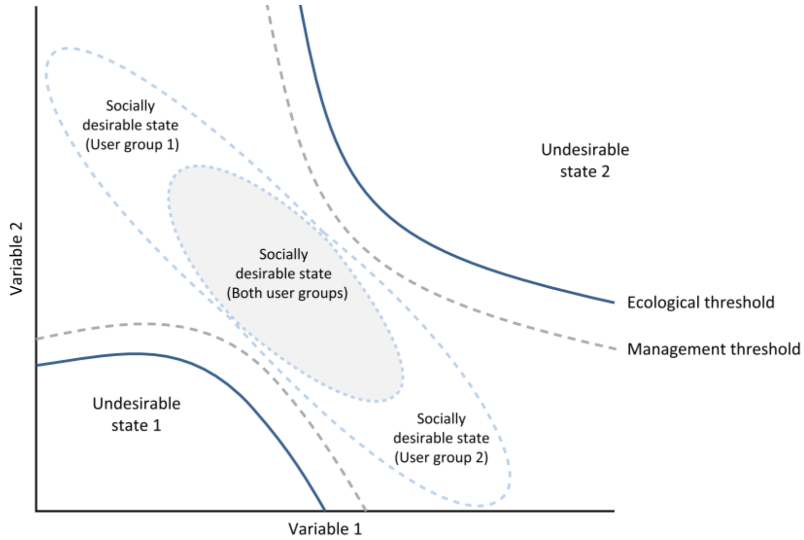



***Figure 2.6: Conceptual diagram showing the interaction between system states, ecological thresholds, management thresholds and stakeholder preferences. (Gillson and Marchant, 2014; p.322).***

Thus, TPCs are informed by current knowledge and understanding of the system since they are locally relevant and appropriately calibrated to the modern variables under study, and according to both ecological variability and stakeholder needs, to ensure adaptive ecosystem-based management. Therefore, using palae-records and PSD to understand feedbacks and trajectories might help us to identify potential thresholds even where we have not encountered them. As an example, the VANG record from Elandsberg PNR where the increasing abundance of Renosterbos pollen allowed for the identification of potential future thresholds where more fire-sensitive (Thicket patches) and grazing-sensitive elements (other Asteraceous shrubs) could be lost and vegetation structure homogenised by increasing unpalatable Renosterbos (Forbes et al., 2018).

### 2.5.3 Useful end-products and capturing learning processes

Boundary crossing involves the effective integration of diverse knowledge types and domains while creating new knowledge through collaborative networks and processes (Akkerman and Bakker, 2011; Jean et al., 2018). The interactions and transitions between stakeholder groups across different sectors and disciplines needs to be considered in applied palaeoecology that informs sustainable development. Thus, it is the role of the applied palaeoecological Community of Practice (CoP) to take the lead in demonstrating the benefits derived from including palaeoecological proxies for ecosystem services within existing ecosystem-based management approaches (O'Higgins et al., 2020). In order for this to occur, novel statistical and modelling methods need to be included in palaeoecological studies so that long-term data can be translated into a usable format (e.g. end-products such as models used as boundary objects) to inform policy and that offers decision-support for landscape and ecosystem managers (Jeffers et al., 2015). In addition to packaging palaeo-data in user-friendly formats or end-products, Therefore, documenting and sharing the novel methodological adaptations they employ, and associated research process steps needed for case-specific and multi-sector research is an essential knowledge management practice needed to progress this field in a variety of South African geographical contexts. As applied palaeoecology uses techniques to gather empirical data that allows us to reflect on the past and develop insights for the present and future and is therefore ‘reflexive’ by nature, the applied palaeoecological CoP needs to consider the benefits of reflective learning (Clifford-Holmes, 2015; Sterman, 2001; Taylor et al., 2016) and the dissemination of insights on the process steps of how we engage with stakeholders and why. Therefore, it is imperative that the applied palaeoecological CoP actively coordinates with governance mechanisms at multiple, adaptive levels of governance (global, national, regional and local scales). The relevance for mainstreaming applied palaeoecology, building meaningful relationships with multiple stakeholders and developing useful end-products is timely in light of sustainable land-use management in the CFR. As there is national and global support for the implementation of ecosystem-based management and governance, including ecological restoration efforts and Ecosystem-based Adaptation (EbA) (DEA and SANBI, 2016; DEFF, 2019; Gann et al., 2019; O’Higgins et al., 2020), it is therefore a potential leverage point to enhance the results and outcomes of collaborative and applied approaches that can activate systemic change, such as the proposed conceptual meta-framework.

# 3 CHAPTER 3: STUDY FOCAL AREA

This chapter provides context for the present study. Ecological infrastructure (mountain catchment, wetlands, corridors of indigenous veg) of the three case study sites within the Berg River Catchment co-produce ES (Cumming et al., 2014; Fischer and Eastwood, 2016). It includes a description of the geology, climate, topography, vegetation and history (i.e. land-use and climatic) of the three study sites under investigation (Groot Winterhoek Wilderness Area (GWWA), Rhenostervlei Farm and Elandsberg Private Nature Reserve (Elandsberg PNR). Important features are illustrated via GIS maps and a historical timeline is constructed for the Middle Berg River Catchment in the CFR. This section describes the reason for selecting the coring sites in terms of the social-ecological context of each site.

## 3.1 The Berg River context

### 3.1.1 Climate and biophysical details of the Berg River Catchment

The Berg River Catchment is located north-east of Cape Town. The entire catchment is located within the CFR of South Africa and is the largest catchment in this region (Figure 3.1). It is a significant water source which supplies water for the metropolitan area of Cape Town and surrounding agricultural lands and rural towns. The Berg River catchment is divided into three sub-areas, based on quaternary catchments and geomorphological zones: Upper Berg River (G10A-B); Middle Berg River (G10C-D); and Middle Berg River (G10E-G10M) (Figure 3.1). The headwaters of the Berg River are in the Franschhoek and Drakenstein mountains. The Berg River is approximately 290 km long with a catchment area of about 9000 km$^2$. As the Berg River Catchment is situated in the winter rainfall zone (WRZ) of the CFR, climatic conditions are characterised by Mediterranean-type conditions with warm dry summers and rainfall being at its maximum during the winter season, representing precipitation in the equatorward margin of the westerly wind belt (Chase and Meadows, 2007; Chevalier and Chase, 2015; Haensler et al., 2011).Within the catchment, rainfall occurs along a sharp gradient from the south-eastern upper catchment (>1200 mm per year) to the north-western estuary (<300 mm per year). Mean annual temperature is between 16-18°C (Midgley et al., 2014; Stuckenberg, 2012). River flow is highly seasonal, with low flows of 0.2 m$^3$/s in the upper river and 2.0 m$^3$/s in the lower river during summer (November–February) (De Villiers, 2007). Higher flows of 4 m$^3$/s occur in the upper river and 15 m3/s in the lower river during winter (May–August) (De Villiers, 2007).

The geology of the upland areas of the catchment is dominated by sandstone and quartzites of the Table Mountain Group (Cape Supergroup). The remainder of the catchment is dominated by shales of the Malmesbury and Klipheuwel Groups, and Cape granites. As a result, soils vary noticeably from sandy sediments in the lower catchment to clayey sediments in the middle catchment (Clark and Ractliffe, 2007; De Villiers, 2007; Midgley et al., 2014; Stuckenberg, 2012). Despite the catchment's typically nutrient-poor soils, the mineral content of watercourses increases rapidly as one moves downstream, due to the varied geology within the catchment (De Villiers, 2007). The rich clayey soils that characterise the Middle Berg River Catchment are one of the main reasons for agricultural development in the lowlands.

The Mediterranean-type ecosystem of the CFR consists of four biomes. One of these is the megadiverse Fynbos biome which has two vegetation units within the Berg River Catchment. These are Fynbos (with respective subtypes – Sandstone and Alluvium) and Renosterveld (with respective subtypes - Shale, Granite, Silcrete, and Alluvium) (Bergh et al., 2014). Fynbos vegetation makes up 25.7% of the total area of the Greater Cape Floristic Region (GCFR) (193 311 $km^2$) and is commonly found on sandstone substrates. The other vegetation unit, Renosterveld, covers 16.1% of the GCFR and is commonly found on granite and shale substrates (Allsopp et al., 2014b; Nathan and Scobell, 2012).

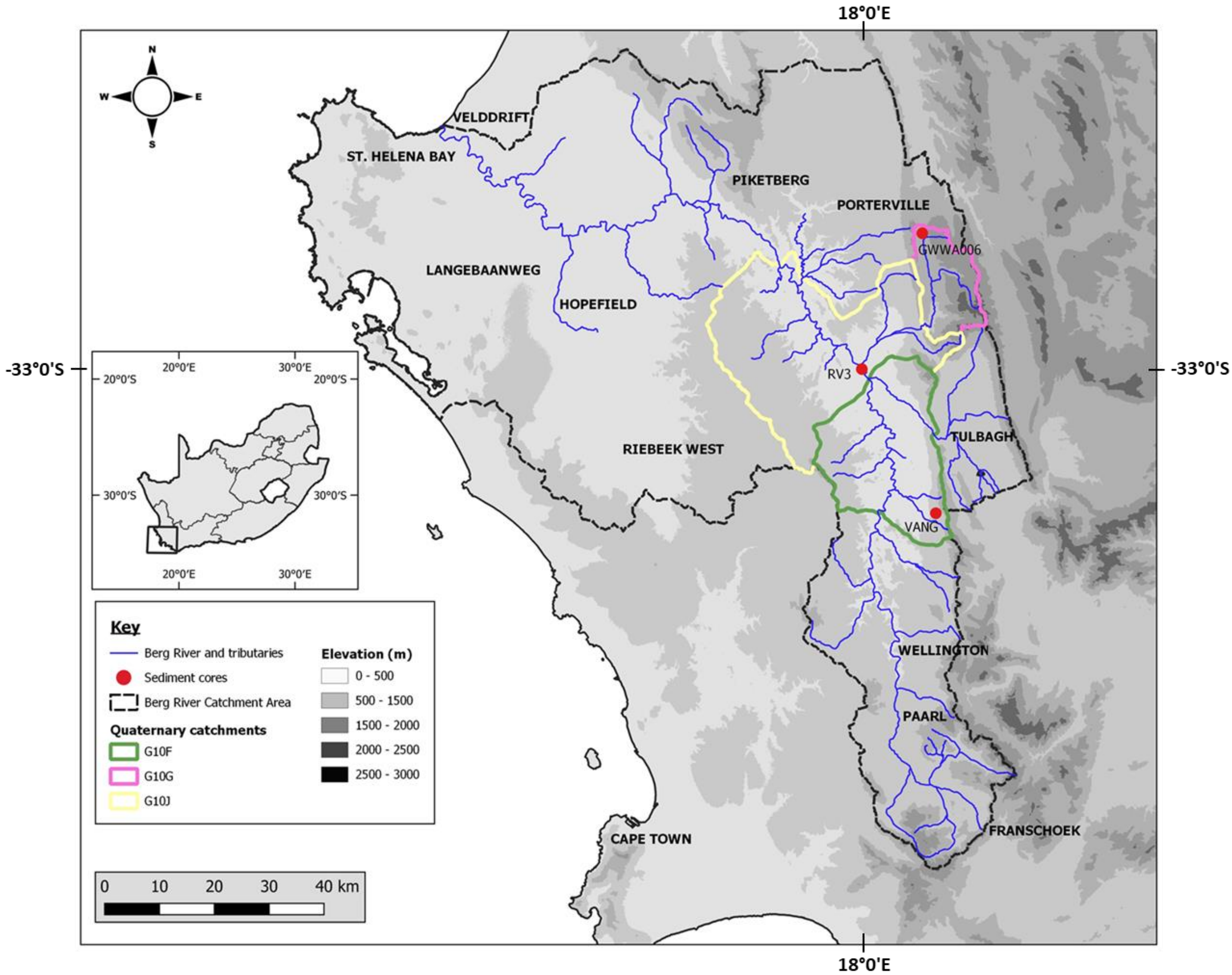



*Figure 3.1: Map showing the general study focal area with three case study sites, respective quaternary catchments, covering an elevation gradient within the Middle Berg River Catchment in the Cape Floristic Region, South Africa.*

### 3.1.2 Land-use and climate history

The CFR has been impacted by climate variability and several different land-use practices for millennia (see timelines of events - Table 5.1 and Table 6.1). For example, San hunter-gatherers were in the region from at least 25 000 BP to the historical period, while Khoikhoi pastoralists introduced livestock to the Western Cape from approximately 2000 BP (Deacon, 1992; Humphreys et al., 1998). Khoikhoi pastoralists would use fire to improve the veld for grazing of sheep and cattle (Deacon, 1992; Smith, 1987). The arrival of European settlers in the mid-17th century resulted in increased grazing, as well as the introduction of crop cultivation and an increase in the use of fire (Meadows, 2003). In the 20th century, agricultural intensification increased environmental impacts in some areas, while others became formal protected areas (private or state-owned nature reserves) and stewardship agreements aimed at conserving the CFR's unique biodiversity (Von Hase et al., 2010). Agricultural intensification characterises the first half of the 20th Century but that from about AD 1970s greater emphasis was placed on protecting the mountains, and this was not only for biodiversity conservation but initially, primarily for water provision (Bergh and Visagie, 1985; Hoffman, 1997; Newton, 2008; van Rensburg et al., 2018; van Wilgen et al., 2016a, 2008).

Over the last ca.10 000 years the CFR has also experienced significant variability in climate. For example, warmer and dryer conditions characterise the Mid-Holocene altithermal (9000-5000 BP) (Meadows and Baxter, 1999) and the Medieval Climate Anomaly (AD 900–1400), while colder and wetter conditions characterise the Little Ice Age (AD 1400–1800) (Nicholson et al., 2013)**.** The latter part of the 20th century has been characterised by a warming trend due to anthropogenic climate change (Cronin et al., 2003; Haensler et al., 2010), and this trend is likely to continue in the region as climate change projections report 30-50% decreased precipitation, 1.5-3.5 °C increased aridity, droughts and extreme weather events (Haensler et al., 2011; Meadows, 2006; Turpie et al., 2002) that poses a significant threat to the region with predictions of dramatic contractions of the Fynbos biome within the CFR by 2100 (Midgley and Thuiller, 2007; Midgley et al., 2005). However, there is some uncertainty in these predictions as other environmental factors besides temperature and rainfall (e.g. evaporative demand and wind) also affect vegetation (Hoffman et al., 2011) and the amplitudes of warming are reported to be underestimated for the subtropics that could experience a 4-6 °C increase in aridity (Engelbrecht et al., 2015). Thus, investigation on the impacts of climate change on vegetation cover response and vegetation-fire dynamics within the spatially heterogenous CFR landscape is complex (Altwegg et al., 2014; Conradie et al., 2022; Slingsby et al., 2017; Van Wilgen et al., 2010).

### 3.1.3 Current threats to ES

The Berg River Catchment provides important provisioning (e.g. water and food production), supporting (e.g. biodiversity, habitat, soil formation), regulating (soil erosion regulation, climate regulation), and cultural services (recreation, education, stewardship) (see section 1.1.1. and Table 1.1). Current threats to ecosystem services in the CFR are the result of the conversion of natural vegetation for urban development and agriculture. The spread of invasive alien plants, inappropriate fire regimes and anthropogenic climate change pose additional threats to biodiversity and other ecosystem services (Allsopp et al., 2014a; Goldblatt and Manning, 2000; Myers et al., 2000; von Hase et al., 2003). In particular, the Berg River Catchment has experienced extensive transformation, primarily for agricultural activities. According to the *State-of-Rivers Report* for the Berg River system, approximately 60% of the area comprises of agricultural land (River Health Programme, 2004). This includes ca. 7% irrigated crops such as grapes and deciduous fruits which are cultivated in the eastern regions, and ca. 53% dryland crops (grains such as wheat and canola) with extensive livestock farming (cattle and sheep) in the west. Forest plantations cover less than 1%, urban areas comprise ca. 2.5% and the remaining 36% is classified as “natural” vegetation cover (River Health Programme, 2004).

The Berg River has experienced major changes over the last century. The Klein Berg tributary, which is upstream from the Rhenostervlei Farm (Figure 3.1), experiences no flow in its lower reaches during summer months yet it was historically perennial (De Villiers, 2007; River Health Programme, 2004), Decreased river flow is closely linked to numerous dams being constructed along the Berg River since AD 1947 and further diversion of water to Voëlvlei Dam since AD 1952 to support agriculture and urban development. Agricultural intensification and urban development, with the spread of invasive alien plants (IAPs) as a by-product, have placed significant pressure on ecosystems and the services they provide. Given current economic activities mentioned above, the Berg River Catchment is heavily dependent on water resources with an annual water demand of 690 Million $m^3$/year. Pegram and Baleta (2014) reported the following distribution: 52% for households and businesses and 43% for irrigation. The remaining 5% of water is utilised by alien vegetation, which covers ca. 155 000 ha of the Berg River Catchment (Cheney et al., 2018; Midgley et al., 2014; Pegram and Baleta, 2014; van Wilgen et al., 2008). Interestingly, Pegram and Baleta (2014) do not comment on the amount of water used by indigenous vegetation within the catchment.

The relatively recent conversion of natural vegetation to agriculture and settlements has resulted in a high abundance of threatened plant species in a fragmented landscape. In addition to the threats on plant biodiversity, there is a notable decrease in water quality in a downstream direction. Higher water quality is evident in the headwaters (e.g. the upland conservation site, GWWA) of most of the tributaries of the Berg River Catchment, with decreased water quality (due to pollution) where the rivers pass through urban settlements and intensive agricultural lands (Midgley et al., 2014). Moreover, the middle to lower reaches of the Berg River has displayed an increasing trend in salt levels since the 1960s (DWAF, 1993), with land-use managers mentioning that water salinity is increasing upstream because of evaporation due to climate warming (multi-stakeholder engagement workshop, July 2019). Although the Berg River is naturally saline due to the geology (section 3.1.1), the increasing trend has been attributed to clearing of indigenous vegetation, Fynbos and Renosterveld, for cultivation. This is known as dryland salinization where agricultural activities alter the water balance releasing stored salts in the top 10 m, which is a common phenomenon in the Western Cape lowlands (Midgley et al., 2014; River Health Programme, 2004). The extent of climatic and land use changes that will affect ecosystem services over time is uncertain, both in the past and future.

## 3.2 The Berg River Catchment showing the focal study area

The present study uses the Middle Berg River Catchment in the south-western portion of the Cape Floristic Region (CFR) (Figure 1.1 and Figure 3.1) to understand the long-term changes in select ES: biodiversity, water quality and soil erosion regulation in response to drivers such as herbivory, fire and climate over decadal-centennial-millennial timescales. Linking with SD terminology, three case study sites within the study focal area were chosen for investigation for ‘boundary setting’ purposes (Brent et al., 2017b). The sites are along a gradient of elevation and land-use intensity: the Groot Winterhoek Wilderness Area (GWWA) with GWWA006 sediment core, Rhenostervlei Farm with RV3 sediment core and Elandsberg Private Nature Reserve (Elandsberg PNR) with VANG sediment core (original data published in Forbes, 2014; Forbes et al., 2018).

### 3.2.1 Upland conservation case study site: The Groot Winterhoek Wilderness Area (GWWA) and associated sediment core, GWWA006

#### *3.2.1.1 The geology, climate and vegetation of GWWA*

The Groot Winterhoek Wilderness Area (GWWA) (-32.975792 S, 19.054922 E) is a protected mountain catchment about 30 608 ha in extent (Holden, 2017; River Health Programme, 2004), situated in the south-western portion of the CFR, ca.15 km north of Tulbagh and ca.4 km east of Porterville (Figure 3.2). A portion of GWWA land (ca.10 000 ha/100 km$^2$) has been under strict protection since AD 1910 whereas another portion (ca. 20 000 ha/200 km$^2$) was privately owned until AD 1978. GWWA was later proclaimed a formal protected area in AD 1985 under the Forest Act of 1968 and it was declared one of the eight protected areas constituting the Cape Floristic Region World Heritage Site (Holden, 2017). The geology of the Groot Winterhoek Mountain, including the GWWA, is typical of the Cape Folded Belt of the Table Mountain Group, predominantly composed of sandstone-quarzitic soils with a subsidiary composition of shale or partly shale-derived soils near the shale-band areas. The annual rainfall ranges from 353- 1776 mm with a mean of ca.742 mm. Areas above 900-1300 m above sea level (such as the area where GWWA006 core was retrieved - Table 3.2) receives substantial amounts of fog and mist throughout the year. During winter, a large quantity of precipitation is converted into streamflow, whereas summer months are characterised by low precipitation, high potential evapotranspiration and conditions of low streamflow and soil moisture (Holden, 2017). The soils of the Groot Winterhoek Mountains support several vegetation categories (Figure 3.2). The main vegetation type in the GWWA, which comprises 91.2% of the area, is Winterhoek Sandstone Fynbos (FFs5) comprising closed “restioland”, proteoid and ericoid Fynbos on higher slopes and asteraceous Fynbos and Cape Thicket/Forest on lower slopes with rocky outcrops (Mucina and Rutherford, 2006). This vegetation type dominated the site where the GWWA006 sediment core was retrieved.

#### *3.2.1.2 Land-use activities of the GWWA and its surrounds*

In comparison with the relatively fertile lowlands of the Middle Berg River Catchment, the generally rocky and rugged GWWA has infertile soils that are not suitable for extensive cultivation, although some areas supported agricultural activities before AD 1978 (Figure 3.2). The most important land-use activities before proclamation in AD 1985 included the cultivation of orchards and fodder crops, livestock farming (sheep, goats and cattle) and the harvesting of indigenous vegetation (Protea and Buchu species). These activities have been associated with European settler agriculture since the late 17$^{th}$ Century (see Table 5.1) for detailed timeline of historical events). Prior to this, the main land-use impacts included the use of fire by Khoi-San hunter/herders who burnt the veld to stimulate the growth of edible indigenous vegetation. Fire was also used to improve the veld for grazing by sheep and cattle and to attract game for hunting. GWWA is currently used for conservation and is a popular ecotourism site, although it has been closed to the public since AD 2016. This closure has been implemented to assist vegetation recovery negatively impacted by unplanned fires (CapeNature, 2016) and erosion (Wheeler pers. Comm. 2016). The GWWA is an important conservation area due to the species-rich Winterhoek Sandstone Fynbos vegetation and its associated fauna (e.g. klipspringer, grey rhebok, leopard and grysbok). A number of rare, threatened and endangered plant species are found at GWWA, such as the flat-leaf clusterhead protea (*Sorocephalus scabridus*) and the 2 m tall shrub, *Ixianthes retzioides* (Stilbaceae), which is endemic to the GWWA’s mountain streams (River Health Programme, 2004). Land-use activities in the uplands surrounding the GWWA today mostly include the cultivation of indigenous plants such as Proteas for the cut flower industry, blueberry farming, ecotourism and private recreation (Holden, 2017). The area adjacent to the GWWA006 coring site in the north, Zuurvlakte Farm, is privately-owned land used for berry farming.

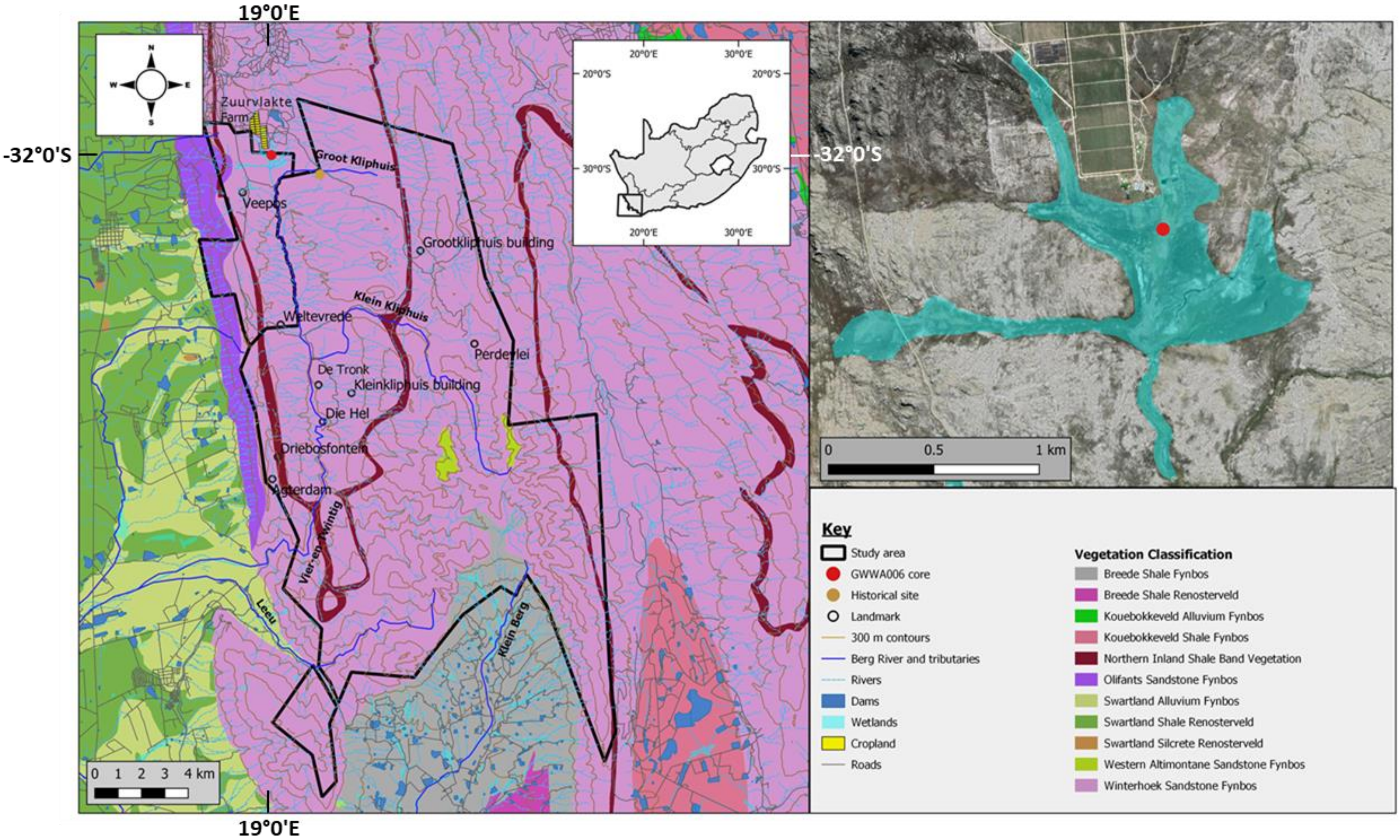



*Figure 3.2: Map of upland conservation site, Groot Winterhoek Wilderness Area (GWWA), within the Middle Berg River Catchment (marked with a box), of the Cape Floristic Region, South Africa. Sediment core GWWA006 (red dot) was retrieved in October 2016 and used for high resolution, multi-proxy palaeoenvironmental analysis in the present study. Google earth image on top right shows the extent of the wetland, and the adjacent croplands to the north of the GWWA. See Figure 5.1 for further site-specific results related to vegetation survey and local topography.*

### 3.2.2 Lowland agricultural case study site: Rhenostervlei Farm and associated sediment core, RV3

#### *3.2.2.1 The geology, climate and vegetation of Rhenostervlei Farm*

Rhenostervlei Farm (-33.227499 S and 18.920272 E) constitutes 539.95 ha of privately-owned land. It is situated in the lowlands of the CFR, ca. 20 km north-east of Riebeek West and ca. 7 km south-west of Saron (Figure 1.1). It is positioned in the lower to middle reaches of the Middle Berg River Catchment. The coring site (RV3) is situated on a floodplain adjacent to the Berg River (Figure 3.3), upstream from the Klein Berg tributary and downstream from the Vier-en-Twintig Riviere tributary.

The geology of Rhenostervlei Farm is composed of the Porterville Formation (of the Malmesbury Group derived 575-540 Ma) on the east of the property and older Moorreersburg Formation (of the Swartland group derived >575 Ma) on the west. As a result, the soil composition is a combination of clayey soils in some areas and more sandy soils (acidic tertiary, grey regic sands, usually white or yellow in colour) in other areas (Mucina et al., 2006). The soils of Rhenostervlei Farm support Atlantis Sand Fynbos (FFd 4) and Swartland Shale Renosterveld (FRs 9) (Mucina and Rutherford, 2006). Renosterveld is the dominant vegetation sub-type closest to where the RV3 sediment core was retrieved, although in recent decades this vegetation has been transformed by agriculture and the spread of IAPs (Figure 3.3). Renosterveld is an evergreen, fire-prone Mediterranean-type or Asteraceous shrubland, which is mostly restricted to fertile, fine-grained soils. Renosterveld comprises of shrubs, grasses and a high diversity of endemic and threatened geophytes (bulbs). An evergreen, unpalatable Asteraceous shrub species, *Elytropappus rhinocerotis* (hereafter, *E. rhinocerotis*) commonly known as Renosterbos, is dominant in many remaining fragments and often categorized as an undesirable shrub.

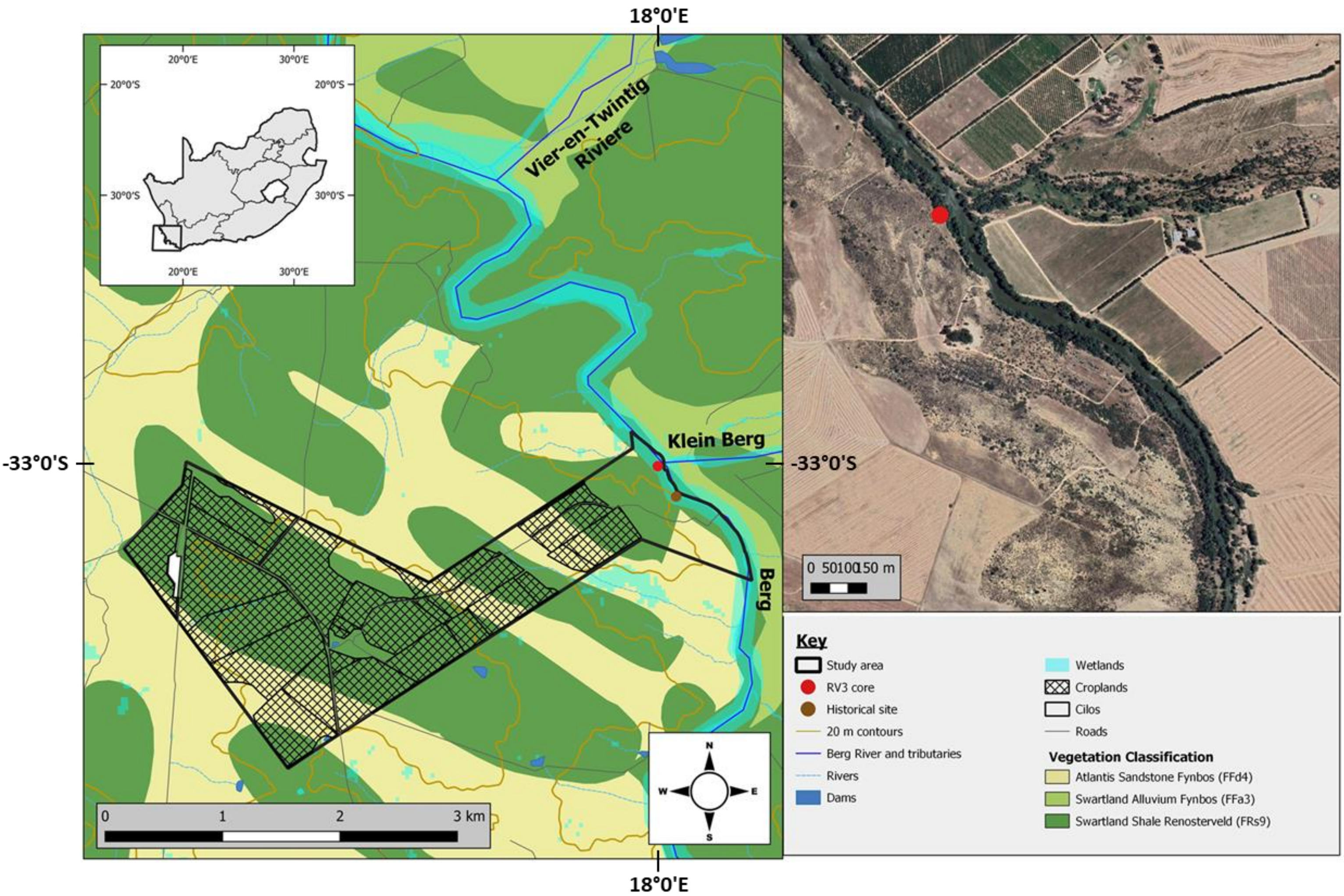



***Figure 3.3: Map of lowland agricultural site, Rhenostervlei Farm, within the Middle Berg River Catchment (marked with a box), of the Cape Floristic Region, South Africa. Sediment core RV3 (red dot) was retrieved in September 2011 and used for palaeoenvironmental analysis in the present study. Google earth image on top right shows the extent of the indigenous vegetation fragments and the surrounding croplands. See Figure 6.1 for further site-specific results related to the vegetation survey and local topography.***

#### *3.2.2.2 Land-use activities of Rhenostervlei Farm and its surrounds*

In the CFR, Renosterveld vegetation has been significantly impacted on by agriculture, with 91-97% of the vegetation transformed (Rouget et al., 2003b; von Hase et al., 2003). This vulnerability of lowland Renosterveld is largely due to the desirability of fertile, nutrient-rich clayey soils that have encouraged the extensive agricultural development seen in the Berg River Catchment today. Rhenostervlei Farm has been managed by the Gildenhuys family for three generations with the original purchase of Rhenostervlei Farm in AD 1906 (Table 6.1). Knowledge of land-use management at this location before the early AD 1900s is mostly undocumented. However, prior to ca. AD 1960, farming practices were considered less intensive as they involved the use of donkeys as opposed to the heavy agricultural machinery currently in use (Gildenhuys pers. comm. 2019). However, rotational grazing and crop cultivation with minimal to no burning is a practice that has been implemented by land-use managers for decades at this lowland agricultural site.

About 81% (439.5 ha) of the total Rhenostervlei Farm area is used for the cultivation of canola, wheat and oats (Figure 3.3). The remaining 19% (100.4 ha) comprises natural vegetation, which is often grazed by cattle and is invaded by alien vegetation. The Rhenostervlei Farm is part of the few farms that contribute to the commercial production of Waterblommetjies in the CFR. Waterblommetjies (commonly known by the English name, Cape pond weed - *Aponogeton distachyos* L.f.) are also grown as a crop in several small dams on Rhenostervlei Farm. Land-use activities surrounding Rhenostervlei Farm mostly include cultivation of wheat, maize, triticale, citrus, plums, wine grapes, and Lucerne (grazing pasture) and livestock (cattle and sheep) farming. Farmlands in the area, like Rhenostervlei Farm, rely on water supplies from the Western Cape Water Supply System, with direct abstraction from rivers as well as small farm dams (Figure 3.3) contributing significantly to their irrigation needs. Commission of the Voëlvlei Dam in AD 1952 regulated water supply in the lowlands. In the decades that followed, changes in water supply (due to climate variability and dam construction) influenced land-use practices and ecological processes at Rhenostervlei Farm, particularly the wetland where RV3 core was retrieved.

### 3.2.3 Lowland conservation case study site: Elandsberg Private Nature Reserve (PNR) and associated sediment core, VANG

#### *3.2.3.1 The geology, climate and vegetation of Elandsberg PNR*

Elandsberg PNR (3500 ha) is a Stewardship Contract Nature Reserve situated on Farm Bartholomeus Klip (-33.45000 S and 19.05000 E) in the Cape lowlands. Elandsberg PNR is approximately 15 km east of Hermon and 25 km north of Wellington (Figure 1.1). The geology is composed of terrace gravel and alluvial debris, with patches of Malmesbury shales in the centre (Mucina and Rutherford, 2006). The soil composition is characterized by sandy lithosols and coarse sand, especially on the steep slopes of the eastern part of the reserve, where the soils are poorer in nutrients (Figure 3.4). The infertile parent rock (Ordovician sandstones of the Table Mountain Group) and the water seepage on the steep slopes is the reason for the leached soils with low nutrient content. However, as the parent rock material further west on the property is Malmesbury shale, the lower slopes and flat areas have more fertile deeper loam soils and fewer rocks (Wooding 2011). Elandsberg PNR has an elevation of 68-400 m above sea level, with the mountains bordering the east side of the reserve reaching heights of up to 1 378 m above sea level. MAP is approximately 600 mm per year, with higher rainfall levels at the top of the Elandskloof Mountains (1000-1400 mm from north to south) (Wooding, 2013). Heavy fog events and increased run-off from the mountain during the winter months are factors resulting in above normal levels of soil moisture that would be expected from local rainfall alone (Newton, 2008; Wooding, 2013). The Mediterranean-type climate and geology supports the following vegetation sub-types: Swartland Shale Renosterveld (FRs 9), Swartland Alluvium Fynbos (FFa 3) and Hawequas Sandstone Fynbos (FFs 10) (Figure 3.4).

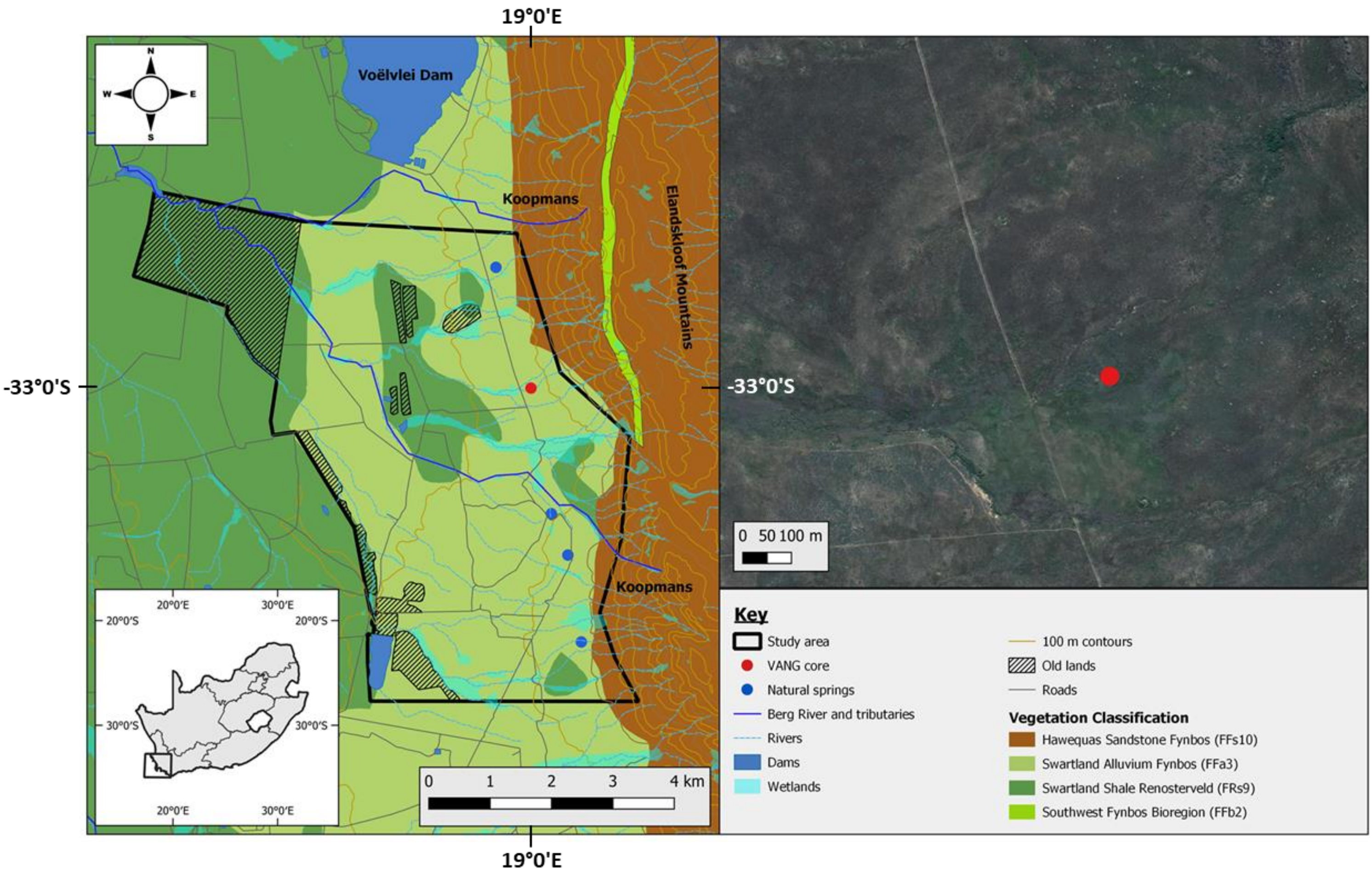



***Figure 3.4:** Map of lowland conservation site, Elandsberg PNR, within the Middle Berg River Catchment, of the Cape Floristic Region, South Africa. Sediment core (VANG) was retrieved in October 2012 (palaeoecological findings published in Forbes et al. 2018) and data used for system dynamics analysis in the present study.*

#### *3.2.3.2 Land-use activities of Elandsberg PNR and its surrounds*

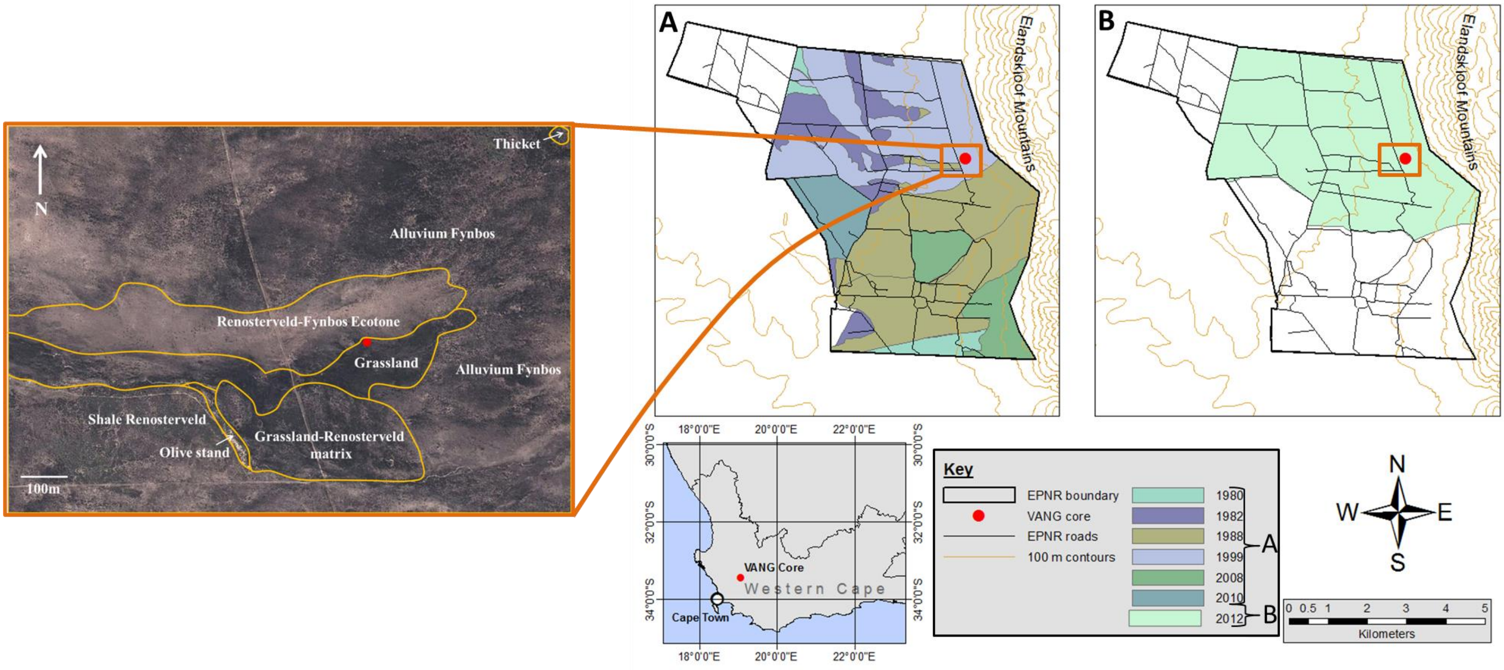



***Figure 3.5:** Google earth map from 2013 shows demarcated areas (in yellow) that represent vegetation units/sub-types that were chosen according to the most dominant vegetation in the vegetation survey. Maps showing fire history seen in (A) from 1980-2010 and (B) the most recent fire in 2012. Roads are included since they often act as fire breaks. Figure adapted from Forbes et al. 2018.*

Elandsberg PNR is an important conservation site since it contains one of the largest areas (ca. >1000 ha) of intact remaining West Coast Renosterveld and is habitat for the critically endangered and endemic geometric tortoise (*Psammobates geometricus*) (Low and Rebelo, 1996). Its importance is confirmed by the location of this site within the Middle Berg River Catchment as it is located south of Voëlvlei dam and lies between highly transformed, fragmented and productive agricultural land in the west and the more protected Elandskloof Mountains in the east. As Elandsberg PNR is generally stony, it is not suitable for intensive cultivation, although there is evidence that some areas supported limited cultivation of wheat and other grain crops before proclamation in AD 1973 (i.e. old lands; Figure 3.4). Land-use activities of the early AD 1900s included grazing by cattle and sheep (Becker, 1996), with palaeoecological evidence showing a greater grazing impact on vegetation than previously thought for this location (Forbes et al., 2018). After AD 1973, large indigenous herbivores (LIHs), which were thought to have lived in this region, were reintroduced into Elandsberg PNR. These included eland (*Taurotragus oryx*), blue wildebeest (*Connochaetes taurinus*), black wildebeest (*Connochaetes gnou*), zebra (*Equus* spp), red hartebeest (*Alcelaphus buselaphus*), gemsbok (*Oryx gazelle*), bontebok

(*Damaliscus dorcas dorcas*) and springbok (*Antidorcas marsupialis*) (historical data shown in Appendix Table 10.8).

### 3.3 Theoretical and practical considerations in selecting the case study sites and respective sediment cores

Analysing multiple sites allows for the consideration of the different socio-economic and ecological benefits each site provides within a multifunctional landscape. The three study sites within the present investigation have varying degrees of protection. These range from formal protection (GWWA and Elandsberg PNR) to no formal protection (Rhenostervlei Farm). The degree of protection potentially influences the levels of disturbance (drivers of change) and therefore the impacts on ecosystem services over time. Such factors as well as the theoretical and practical considerations need to be taken into account when choosing a suitable core.

South Africa is a semi-arid country as it receives approximately 450 mm Mean Annual Precipitation (MAP), which is below the world’s 860 mm MAP (Botai et al., 2018; DWAF, 2004; Shand and Basson, 2003). The paucity of natural lakes and low MAP resulting in a few perennial wetlands means that palaeoecological research to date is relatively scarce, particularly within the Middle Berg River Catchment (see noteworthy palaeo-sites in Figure 2.2a and other sites that could be analysed and cored in Figure 2.2b). As rainfall is strongly seasonal in the winter rainfall zone (WRZ) of the CFR (Figure 1.1), the wetlands are often ephemeral, subject to potential loss of surface material during periods of desiccation and therefore the lack of a continuous stratigraphic record. However, palaeoenvironmental studies from such ephemeral wetlands across much of southern Africa, locally referred to as pans or vleis within the wetlands classification systems (de Klerk et al., 2016; Ollis et al., 2015) remains largely unexplored (Rose et al., 2021) yet are an appropriate source for obtaining interesting palaeoecological results within the CFR (Forbes et al., 2018; MacPherson et al., 2018). Furthermore, to ensure good pollen preservation and stratigraphical integrity, only wetlands that have not been heavily degraded by dredging or significant disturbance by animals are suitable for analysis. For good pollen preservation, wetlands should be anoxic, acidic and waterlogged (Bennett and Willis, 2002).

In addition to the topography of the landscape, the wind speed and direction, and the size of the pollen grains (Hill, 1996; Hill and Finch, 2021), consideration when choosing suitable coring locations is the basin size of the wetland, as this plays a role in determining the pollen source area (Jacobson Jr and Bradshaw, 1981; Sugita, 1994). The source area increases as the basin size increases, where large wetlands collect pollen from a larger area compared to smaller wetlands that reflect a more local pollen signature (see Table 3.1 and Table 3.2 for general and site-specific information, respectively). Typical patch-mosaic landscapes of the CFR have small sediment basins (radius = 50 m or less) and ca. 50% of the pollen would originate from the 300-400 m surrounding the basin, thus local to extra-local scale (Sugita, 1994).

***Table 3.1: Relationship between the size of the site cored and the pollen source area.***

| Reference | Diameter (m) | Area ($m^2$) | Pollen source (m) |
|---|---|---|---|
| | <100 | 7854 | 20 (local) |
| | 100-300 | 7854-70 686 | 20 to several 100 (extra-local) |
| Jacobson and Bradshaw (1981) | >300 | >70 686 | regional |
| | <4 | 12.6 | 50-100 |
| | 100 | 7854 | 300-400 |
| Sugita (1994) | 500 | 196 350 | 600-800 |

***Table 3.2: Information about the locations of sediment cores investigated within the focal area, the Middle Berg River Catchment.***

| **Name of case study site** | Groot Winterhoek Wilderness Area (GWWA) | Rhenostervlei Farm | Elandsberg Private Nature Reserve (Elandsberg PNR) |
|---|---|---|---|
| **Land-use classification in the present study** | Upland conservation site | Lowland agricultural site | Lowland conservation site |
| **Quaternary catchment (source: Water Research Commission)** | G10G | G10J | G10F |
| **Type of analysis in the present study** | Palaeoenvironmental | Palaeoenvironmental | SD modelling |
| **Name of sediment core** | GWWA006 | RV3 | VANG (Forbes, 2014; Forbes et al., 2018) |
| **Wetland location: co-ordinates of coring site** | 32°59' 6.072" S and 19° 4' 14.16" E | 33°12'54.60"S and 18°56'57.60"E | 33°26'17.94"S and 19°4'8.28"E |
| **Wetland elevation: Metres above sea level** | 959 m | 46 m | 151 m |
| **Wetland basin diameter** | ±120-500 m diameter | ±16-28 m diameter | ±22.5 m diameter |
| **Wetland basin area** | 85 144 $m^2$ | 200-615 $m^2$ | 397.6 $m^2$ |
| **Estimated pollen source area** | 20-800 m (extra-local to regional) | 20-100 m (local to extra-local) | 20-100 m (local to extra-local) |

With these practical and theoretical considerations in mind several potential sites were identified for coring. After initial subsampling and conducting a preliminary pollen analysis of a subset of the cores, two undisturbed sedimentary sequences, GWWA006 and RV3 (Table 3.2) were selected for high temporal resolution, multi-proxy palaeoecological and geochemical analyses. Detailed methods are described in section 4.1.3.2 of Chapter 4. The third sedimentary sequence from Elandsberg PNR (VANG core) was analysed using palaeoecological techniques in a previous study (Forbes, 2014; Forbes et al., 2018). However, the palaeoecological data from this site was used for original system dynamics (SD) modelling research in the present study. Choosing these three case study sites within a focal area like the multifunctional landscape of the Middle Berg River Catchment, can generate valuable insights for land-use management decision-support and policy in the CFR.

## 3.4 The social-ecological systems (SESs) of the Middle Berg River Catchment

### 3.4.1 GWWA – upland conservation case study with limited land-use intensity

The Groot Winterhoek Mountains is a poorly studied region, mainly due to accessibility (Mucina and Rutherford, 2006). The area is remote and the rocky terrain can make it difficult to navigate. Until a recent study focusing on the impact of mountain protection on land-use/cover, vegetation, fire and streamflow over the last ca.50 years (Holden, 2017; Holden et al., 2021), there was no focused research on these topics conducted in the area since the AD 1980s and there has been no previous palaeoecological research. Therefore, a palaeoenvironmental investigation is paramount for the understanding of long-term changes in ecosystem services and drivers to support land-use management. Despite the perceived difficulties in accessibility, this upland conservation site, GWWA, has a wealth of wetlands suitable for coring. These wetlands were identified by consultation with the reserve managers and expert ecologists. Although nine wetlands were identified as potential coring sites during a reconnaissance, only six were cored successfully (Figure 2.2), and are stored at the University of Cape Town for possible future palaeoecological analysis. One of these sedimentary cores (GWWA006 - Figure 3.2) was selected for further analysis based on the wetlands location in the landscape, the basal date of the core and the perceived level of land-use disturbance in the GWWA upland conservation site.

Although the GWWA has been used for agriculture in the past (Table 5.1), the area is believed to have experienced lower levels of disturbance. Its remote location and low soil fertility makes it less desirable for intensive farming compared to the case study sites in the lowlands. GWWA has been formally protected since AD 1985, so land-use disturbance has been low in the last three decades. However, given the GWWA006 core's proximity to Zuurvlakte Farm (ca. 200 m north) (Figure 3.2), it is possible that a farming signal would be detected in the palaeo-record. GWWA is managed by CapeNature, a public institution also known as a government parastatal responsible for biodiversity conservation in the Western Cape. CapeNature is governed by the Western Cape Nature Conservation Board Act 15 of 1998.

CapeNature is one of the primary stakeholders in this study, particularly the GWWA reserve managers. As conservation practitioners, they offer valuable insights into changes in ecosystem services and drivers that are important for the sustainable governance and management of land-use in the CFR. Their objective 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, 2016). In view of CapeNature's Biodiversity Stewardship Programme, which aims to support private landowners in promoting responsible land use management, CapeNature can be viewed as a strategic partner for further research and implementation of the protection of biodiversity. Stewardship programmes such as this play a critical role in bridging the gap between the conservation and agriculture sectors.

### 3.4.2 Rhenostervlei Farm – lowland agricultural case study with heavy land-use intensity

In comparison with the GWWA, the lowlands are easily accessible and are therefore more heavily used and thoroughly researched than more remote areas (Cullis, 2018; Currie et al., 2009; De Villiers, 2007; Dzikiti et al., 2016; Molebatsi, 2019; Nyemba, 2013; Stuckenberg, 2012). A number of important regional plans and programmes have also been implemented. They include, the Berg River Improvement Plan (BRIP) that commenced in 2013 to address factors influencing river water quality (i.e. pollution from urban settlements, wastewater effluent discharges and agricultural runoff) (Horn, 2020); the Food, Energy, Water, Land and Biodiversity (FEWLB) Nexus framework that considers the demands on shared natural resources and emerging constraints to local economic development (Midgley et al., 2014); and the Department of Water and Sanitation's recent efforts in determining water resources classes and associated resource quality objectives for the Berg River Catchment (Cullis, 2018). Despite extensive literature on the Berg River Catchment, there is minimal palaeoecological evidence from the area (Figure 2.2), reiterating the importance of this lowland site and relevance of the present study.

Regardless of the easier accessibility in the lowlands, there are very few wetlands that are appropriate for palaeoenvironmental research since many are degraded, do not accumulate enough sediment for a temporally appropriate analysis, or the sediment does not preserve pollen or diatoms well. However, several wetlands were identified (Figure 2.2b) by consultation with farmers, researchers from Elsenburg Agricultural Training Institute and expert ecologists from CapeNature, with one sediment core (RV3) eligible for further analysis in the present study based on evidence of intact sediment and pollen preservation. The Rhenostervlei Farm case study area likely experienced higher levels of disturbance than upland sites or those with stricter levels of protection. Furthermore, the coring site (RV3 - Figure 3.3) is situated along the riverine fringe on a floodplain adjacent to the Berg River, upstream from the Klein Berg tributary and downstream from the Vier-en-Twintig Riviere tributary. Consequently, this site receives deposition of nutrient-rich sediment from the river when it floods after heavy rains. This location makes it a good site for analysing the impacts from river modifications undertaken during the period AD 1950s-1970s such as the construction of Voëlvlei Dam and diversion weirs in the Klein Berg and Vier-en-Twintig Riviere rivers that altered flow patterns and potential changes in upstream activities.

Although ca.18% of Rhenostervlei Farm has not been cultivated, areas containing indigenous vegetation are not managed under any of the voluntary options offered under CapeNature's dedicated Biodiversity Stewardship Programme. In addition to indigenous plant biodiversity at the site, historical ruins were discovered in the early AD 1900s, all of which contribute to the area's natural and cultural heritage (Figure 3.3). Since agriculture not only makes a significant contribution to our economic development but also to the co-production of ecosystem services (Bengtsson, 2015; Fischer and Eastwood, 2016), commercial farmers, particularly the land-owner of Rhenostervlei Farm, are primary stakeholders in this study. Additionally, land-use governance institutions are equally important stakeholders. For example, the Western Cape's Department of Agriculture (LandCare) plays a critical role in sustainable land-use and therefore, in bridging the gap between conservation and agriculture. They may be considered as liaison who can act as 'knowledge brokers' as they foster relationships and networks with private landowners, scientists and policy-makers in this region. More than 90% of the Cape Lowlands is privately-owned and falls outside of formally protected conservation areas (Von Hase et al., 2010), therefore, farmers are *de facto* custodians of the CFR's endangered biodiversity. Farmers can also be viewed as strategic partners for further research into and implementation of biodiversity conservation outside of formal protected areas such as nature reserves.

### 3.4.3 Elandsberg PNR – lowland conservation case study with intermediate land-use intensity

Elandsberg PNR is in the Cape lowlands and is easily accessible. It was used for agriculture in the past but is now protected so can be considered intermediate in terms of land-use intensity. There is relatively widespread literature from previous ecological studies conducted at Elandsberg PNR and the adjacent farm (C. B. Krug et al., 2004; R. M. Krug et al., 2004; Midoko‐Iponga et al., 2005; Walton, 2006), but the first palaeoecological study at the site was conducted by Forbes (2014) (Forbes et al., 2018). In consultation with CapeNature, Elandsberg PNR reserve manager and farmers form a part of the Agter-Groenberg Conservancy, three wetlands were identified as coring sites in 2012 (see pink dots in Figure 2.2b). VANG core was chosen for analysis based on the small size of the closed basin, pollen preservation and the basal date, which resulted in a sequence of ca. 1300 years to determine a pre-colonial baseline in an endangered heathland like West-Coast Renosterveld (Forbes, 2014; Forbes et al., 2018). Given the poor soil fertility of the site, which renders it unsuitable for intensive crop production as well as the formal protection since its proclamation as a nature reserve in AD 1973, it was hypothesised that the site had experienced a moderate level of disturbance compared with the highly protected GWWA and the highly transformed Rhenostervlei Farm sites. However, Forbes et al. (2018) reported an unprecedented regime shift to an ecological state dominated by unpalatable Renosterbos and increased grazing and fire since the AD 1950s, likely caused by agricultural intensification at the site. Trends of increased Renosterbos, high fire levels and grazing pressure continued despite changes in land-use after the AD 1970s. Due to its geographical location and moderate level of disturbance over centennial-millennial timescales, Elandsberg PNR as a lowland conservation site within the Middle Berg River is regarded as a suitable case study for applying the proposed conceptual meta-framework (Figure 1.2), and is therefore a proof of concept in blending palaeoecology and PSD (see section 4.2.3.1).

# 4 CHAPTER 4: METHODOLOGICAL APPROACH

The present study is interdisciplinary by design (Allsopp et al., 2019; Elena M. Bennett et al., 2015; Roux et al., 2017; Sellberg et al., 2021). It reconstructs the environmental history of two sites (see Chapters 5 and Chapter 6) within the selected study focal area, Middle Berg River Catchment, in the Cape Floristic Region (CFR) using palaeoecological data. It also addresses land-use disturbance (herbivory and fire) and climate change as drivers of landscape change and interprets the palaeoenvironmental data in terms of ecosystem services such as plant biodiversity, water quality and soil erosion regulation and resilience. Therefore, the present chapter describes the palaeoenvironmental field, laboratory and statistical methods used in sediment core retrieval, multi-proxy data collection and analyses to fulfil Objectives 1-3. A description of the palaeo-proxies for ecosystem services are also described in alignment with relevant literature (Dearing et al., 2012; Jeffers et al., 2015). Finally, the chapter describes the combination of stakeholder expertise (Component B) and a simulation experiment (Objective 4) by way of participatory system dynamics (PSD) modelling to explore feedbacks between the selected ecosystem services, environmental and social-ecological drivers (see Chapter 6), as a basis for land-use management decision-support (Component C).

## 4.1 Palaeoenvironmental methods to reconstruct environmental history (Objective 1)

### 4.1.1 Field methods: sediment core retrieval and collection of sediment surface samples

A mechanical coring device (“vibracorer”) comprising a handle attached to the aluminium pipe and a generator facilitated penetration of sediment with minimal disturbance. An aluminium pipe of 7.62 cm in diameter, 1.6 mm thickness, and between 2-5 m in length was used to core the wetlands. At both coring sites (suitable wetlands at GWWA with GWWA006 core and Rhenostervlei Farm with RV3 core), coring ceased when a coarse sandy layer with gravel-like pieces prevented further coring. A rubber bung was inserted into the top of the aluminium pipe to create a vacuum to keep the sediment in place while the core was retrieved from the wetland. Approximate compression was measured by comparing the depth of the aluminium tube inserted into the sediment (i.e. the total drilling length) with the actual sediment core length. The GWWA006 retrieved sediment core length was 296 cm but only the top 144 cm was analysed for palaeoenvironmental proxies, and the RV3 retrieved sediment core length was 120 cm but only the top 108 cm was analysed for palaeoenvironmental proxies (see section 5.3 and 6.2, respectively, for more details). The decision to analyse the respective lengths of the cores were based on pollen preservation and the temporal scale required to address the research questions.

Excess piping was cut off using a hand-held saw. The exposed ends of the core were packed with plastic bags and sealed with tape. Additionally, a trowel-sized amount of surface sample sediment was collected from the top 0-3 cm close to where the cores were retrieved. Surface samples were collected in order to assess modern pollen, and to establish the relationship between the vegetation type present in the area at the time of sampling and the pollen spectra. Other methods include deploying modern pollen traps (Hill and Finch, 2021) and the collection of large amounts (>5) of soil surface samples (Hill et al., 2021) but neither were used in the present study the calibration of modern-pollen vegetation relationships (e.g. Meadows and Sugden, 1991b, 1990) was outside of the scope. However, the analysis of surface sediment samples is a commonly used method for describing the pollen signature of modern vegetation in terms of diversity or vegetation types (Lézine et al., 2021). The cores and surface samples were measured, labelled, and transported to the University of Cape Town (UCT) for analysis. At UCT, the aluminium pipes were split in half using a bandsaw, with little or no displacement of the sediment. Both halves of the core were wrapped in plastic cling wrap, placed in a black plastic bag and stored in a 4 °C fridge to inhibit the growth of micro-organisms. One half was used for fossil pollen, spores, charcoal, diatom and lithology analyses whilst the other half was used for geochemical analyses and chronology.

### 4.1.2 Field methods: vegetation survey

Surveys of the vegetation surrounding the Rhenostervlei Farm and GWWA coring sites were carried out to assist with the interpretation of the palaeo-records. The vegetation survey included an adaptation of the Braun-Blanquet (BB) approach (van der Maarel, 2007; Westhoff and van der Maarel, 1978) and was structured by distance from the wetland and by patches of different types of vegetation. The sampling was stratified by identifying distinct vegetation or landscape units based on google earth images and the location of the coring site. After selecting the vegetation units, plots were chosen given that the vegetation represented a typical vegetation community and was not on an ecotone/transition to another vegetation unit. The vegetation survey consisted of assessing the vegetation within a plot size appropriate for the vegetation size classes. The survey area was either 10 $m^2$, 25 $m^2$ or 400 $m^2$ and wooden poles were used to mark the four corners of the plot.

The abundance of dominant species (i.e. those with >5% cover) present in each plot was determined by consensus (agreement by three to four people). Although a low threshold of 5% was not required for the interpretation of the pollen records, the adapted BB protocol was based on an area guide for BB relevés which included 1-9 Ordinal Transfer Values (van der Maarel, 2007; Westhoff and van der

Maarel, 1978) that corresponded with percentage abundance ranges within the plot. As the fynbos vegetation is within a mega-diverse ecosystem there are many species in low abundances. Therefore, there may be instances where a BB plot has multiple species from the same family that are in low abundance (0-10%). Thus, recording the lower thresholds of dominance is important when calculating the average estimated dominant vegetation surveyed and interpreting the pollen record. Unidentified types were given a collector's number and photographed and placed in plastic bags for plant identification to family and genus level, and their respective percentage abundances were recorded. GPS coordinates, aspect, elevation, dung counts and bare ground cover were also recorded. Photographs were taken of the plot as a visual reference for the type of vegetation and where it was situated in the landscape. After all replica plots were surveyed within the vegetation unit (i.e. three to four BB relevés), information on the SES context was also recorded. This included the general vegetation pattern within the landscape; climatic and geological conditions, evidence of past fire and fire refugia, geomorphological/hydrological information such as whether sites were infilled stream valleys and controlled by groundwater or runoff, showed evidence of erosion and additional notes on the historical socio-economic context related to land-use of the vegetation unit (see BB datasheet in Appendix 10.2 and SES datasheet in Appendix 10.4.2). Information on the SES context was determined by observations whilst in the field together with contributions from stakeholders during semi-structured interviews conducted during the field work.

### 4.1.3 Laboratory methods

Laboratory procedures which included palaeoecological and geochemical techniques are summarised in flowcharts and tables below with further details described in depth in the text and associated Appendices. The same depths in a given core were analysed for pollen, spores, charcoal, diatoms, composition and physical properties of the sediment and geochemistry to provide high resolution, multi-proxy information about changes in biodiversity, herbivory, fire, water quality, climate change and soil erosion regulation. Sub-sampling was not conducted in a pressure-controlled laboratory at UCT, however, windows and doors were kept closed and standard sterile procedures were followed to prevent contamination from local, modern airborne pollen during palaeo-proxy extraction and analysis.

#### *4.1.3.1 Chronology and age depth modelling of sediment cores*

By following standard procedures, a combination of dating techniques including accelerator mass spectrometry (AMS) radiocarbon dating (Lowe and Walker, 2014) and Lead-210 ($^{210}$Pb) dating sediment (Appleby, 2001; Appleby et al., 1979), were used to establish a chronology for the two cores analysed for palaeo-proxies in this study (see Table 4.1 and detailed procedures described in Appendix 10.3.1). Initially AMS radiocarbon dating was used to determine the basal age of the cores. Samples were taken from the GWWA006 core at 144 cm and the RV3 core at a depth of 108 cm (Table 4.1) by adaptation of the volumetric displacement method (Mooney and Tinner, 2011) using a syringe and razor blades to cut and remove a stratigraphic slice of sediment (2-10 g). The samples were individually packaged and sent to the Beta Analytic Inc. Laboratory in Florida at the North American Facility Headquarters for analysis.

After basal date of the RV3 core and initial pollen analysis at the sub-sample level, where specific exotic pollen types (*Cerealia, Eucalyptus, Pinus* and *Acacia*) were used as markers, it was suspected that the top part of the cores contained sediments from the last 150 years and that $^{210}$Pb dating would be appropriate (see below and Appendix 10.3.1.2). Given the relatively old basal date of the GWWA006 core, an additional six samples (Table 4.1) were sent to iThemba LABS - Laboratory for Accelerator Based Sciences in Braamfontein, South Africa, for radiocarbon dating. $^{210}$Pb dating was not used for GWWA006, because the top 4 cm was dated to ca. 1700 years old. A further sample at 0 cm was radiocarbon dated to confirm this rationale.

The analysis of $^{210}$Pb requires a continuum of sediment to be a maximum of 150 years old or younger (Bennion and Appleby, 1999) and each section of sediment should amount to a minimum of 2 g of wet weight. Given the relatively young basal date of the RV3 core at 108 cm, the core was subsampled in a continuum until 100 cm was reached (Table 4.1). Thinner sections of 2-3 cm were sampled at the top of the core and thicker sections of 5-7 cm lower down the core with the aim of including several levels of sediments that have no measurable quantities of unsupported $^{210}$Pb left (e.g. 90-95 cm, 95-100 cm), as required by the constant rate of supply (CRS) model described in Appendix 10.3.1.2. Samples from the RV3 core were placed in ziplock plastic bags and sent to Core Scientific International in Canada for $^{210}$Pb-dating.

***Table 4.1: Information regarding the samples analysed to determine the chronology of the sediment cores that were retrieved from the upland conservation site, GWWA, and the lowland agricultural site, Rhenostervlei Farm.***

| Name of core | GWWA006 | RV3 |
|---|---|---|
| **Number of depths analysed for AMS radiocarbon** | 8 | 1 |
| **Depths (sections) analysed for AMS radiocarbon dating** | 0 cm, 4 cm, 20 cm, 56 cm, 88 cm, 124 cm, 132 cm, 144 cm | 108 cm |
| **Number of depths analysed for lead-210 dating** | 0 | 33 |
| **Depths (sections) analysed for lead-210 dating** | N/A | 0-30 cm in 2 cm increments, 30-60 in 3 cm increments, 60-100 in 5 cm increments |

The ages of the radiocarbon dated samples were calibrated to give dates in calendar years (AD or cal BP) using a standard calibration programme called CALIB (http://calib.org/calib/) which used the Southern Hemisphere terrestrial samples calibration curve (SHCal20) (Hogg et al., 2020). Calibrated dates were not included in the age-depth models but were determined to provide a reflection of error margins in the Bacon modelled calibrated ages (see below) Furthermore, t the $^{210}$Pb ages were used with an added 1-year error without calibration. The CRS sediment accumulation rate was calculated by dividing the cumulative dry mass at the bottom of the extrapolated sediment section by the calculated age at that depth.

Age-depth modelling was done using the package called Bacon (Blaauw and Christen, 2013) version 2.2 in R (http://www.R-project.org/) (see Appendix section 10.3.9 for R script). Radiocarbon dates with errors were included in the Bacon model. This approach to age-depth modelling uses Bayesian statistics to reconstruct Bayesian accumulation histories for deposits. Bacon was selected over the Clam package since Bacon accommodates outlying dates well, as the dates are modelled using a student-t distribution with wide tails (Andrés Christen and Pérez E, 2009). Bacon combines radiocarbon and other dates with prior information such as the CRS model and Slope Regression model used for the $^{210}$Pb dates in the case of the RV3 core. Error margins for the dating methods are reported in the Appendix Table 10.4 (GWWA006) and Table 10.6 (RV3) and taken into account when employing the Bacon age-depth modelling.

#### *4.1.3.2 Pollen, spore and charcoal extraction and analysis*

Sediment sub-samples (1 cm$^3$) were collected from the core using a 3 ml plastic medical syringe with the tip removed. The use of the syringe for volume measurements is an adaptation of the volumetric displacement method (Mooney and Tinner, 2011). For compacted sediments, packing or cutting a known volume is less time-consuming and as reliable as volumetric displacement and provides a representative sample (Carcaillet et al., 2001). Sub-samples were initially taken at evenly spaced intervals (8-12 cm) down the length of the cores. After range finder radiocarbon and $^{210}$Pb dates had been obtained for the GWWA006 and RV3 cores, respectively (section 4.1.3.1), sampling resolution was increased to 4-8 cm and thereafter 0.5-2 cm in areas of the cores that showed significant changes in palaeo-proxies. Fossil pollen, spores, macro- and micro-charcoal were extracted from sediment sub-samples according to standard procedures which include chemical digestion of unwanted sediment fractions using strong acids and alkalis (Bennett and Willis, 2002; Moore et al., 1991) (Figure 4.1). Details regarding the extraction and analyses of each proxy can be found in Appendix section 10.3.3 (pollen), Appendix section 10.3.4 (spores) and Appendix 10.3.5 (charcoal). Micro-charcoal and spores were identified on the same slides as fossil pollen, while macro-charcoal fragments (>150 µm) were retrieved during sieving and counted separately. Palynomorphs were analysed using a Leica DM750 microscope with 4x, 10x, 40x and 100x objectives, and most identification took place at 400x magnification and sometimes 1000x total magnification in conjunction with an immersion oil. The calibrated stage of the microscope was used to traverse the slide systematically at 1 mm intervals. For each sub-sample depth of the sediment cores analysed, a minimum of 250 terrestrial pollen grains (excluding aquatics and marginal pollen types: Cyperaceae, Juncaceae, *Polygonum* and *Typha/Phragmites*-type) were counted to allow for statistical significance and the calculation of 95% confidence intervals (Barkley, 2009; Hill, 1996; Keen et al., 2014; Maher, 1972; Quick et al., 2011; Trombold and Israde-Alcantara, 2005). A minimum pollen sum of 200 (for three of the 53 depths when pollen abundance was relatively scarce) and maximum of 331, with an average of 290 were counted for the GWWA006 core. A minimum pollen sum of 278 and maximum of 468 was counted for the RV3 core. Unknown pollen grains were photographed, included in the pollen sum and later identified or classified as 'Unknown' in the palaeo-record. When pollen was not identifiable it was classified as either 'Concealed' or 'Degraded' and excluded from the pollen sum.

To facilitate the handling of data for further analyses, fossil pollen data were grouped into types definitive of vegetation within Fynbos and Renosterveld classification to analyse between- and within-Fynbos biome changes in relative abundance over time. These vegetation categories included 'Renosterbos' (as dominant taxon in 'Renosterveld'), 'Renosterveld and Fynbos', 'Fynbos', 'Grassland', 'Thicket/Forest-type' and 'Unknowns'. Exotic pollen types were grouped into a separate vegetation type and these included *Cerealia* and invasive alien plants (IAPs) *Eucalyptus*, *Pinus* and *Acacia*. The vegetation categories followed published classifications of biogeographic and floristic summaries within the Fynbos Biome of the CFR (Bergh et al., 2014; Mucina and Rutherford, 2006; Nathan and Scobell, 2012) as well as the classifications used in previous palaeoecological studies within the CFR (Forbes et al., 2018; MacPherson et al., 2018; Meadows et al., 2010; Meadows and Sugden, 1993; Neumann et al., 2011; Quick et al., 2011; Scott and Woodborne, 2007b, 2007a). Given the long-standing debate about the relative abundance of grasses in comparison to shrublands in the Fynbos biome (Cowling et al., 1986; Curtis, 2013; Curtis and Bond, 2013; Forbes, 2014; Newton, 2008; Thom, 1954, 1952) (see section 2.2.2.2), Poaceae pollen have been kept as a separate vegetation type, 'Grassland'. Furthermore, the theoretical hypothesis proposed by Gillson et al., (2020) of quasi-stable states due to Fynbos internal reorganisation/self-organisation to maintain ecological, functional and structural integrity (see section 2.5.2) is recognised and considered in the interpretation of the original palaeo-data derived from GWWA006 and RV3 cores. This included the consideration of whether grasses should be a sub-type of Fynbos and not a separate biome, Grassland, at the upland and lowland sites. This is regarded as an essential data handling decision to corroborate or deny past findings and, therefore, further investigate the historical composition of Fynbos and Renosterveld landscapes for consideration of reference conditions/states important for conservation management and restoration planning.

For each level a minimum of 100 coprophilous fungal spores (*Sporormiella*, Sordariaceae, *Gelasinospora* and *Coniochaeta*) and 100 *Lycopodium* exotic spores were counted (Ekblom pers. comm. 2013). If coprophilous fungal spores were scarce (although there was still a number of non-coprophilous fungal spores present) and a coprophilous spore sum of 100 could not be reached, methods by Gelorini et al. (2012) were adapted so that counting of coprophilous spores continued until 100 terrestrial pollen grains were encountered, to ensure statistical robustness of the results. Spores which were not coprophilous (i.e. Trilete, Monolete and Other non-coprophilous spores) were also counted and were classified as 'non-coprophilous spores' and therefore included in the total spore sum.

Both pollen or spore concentrations were calculated by using the following equation modified from Bennett and Willis (2002):

$$Fossil\ pollen\ or\ spores\ concentration = \frac{Lycopodium\ added \times fossil\ pollen\ or\ spores\ counted}{Lycopodium\ counted}$$

Micro-charcoal abundance was estimated using a slight adaptation of the point count estimation proposed by Clark (1982).

$$Charcoal\ concentration\ = \frac{f.o.v.\ \times no.charcoal\ hits}{no.points} \times \frac{\frac{Lycopodium\ added}{Lycopodium\ counted}}{sediment\ volume}$$

Macro-charcoal (>150 µm) was counted using a binocular dissecting microscope (Leica EZ4) at 10x magnification. The angular black (opaque) particles were counted. The result was expressed as exact particle counts for each level (Duffin et al., 2008).

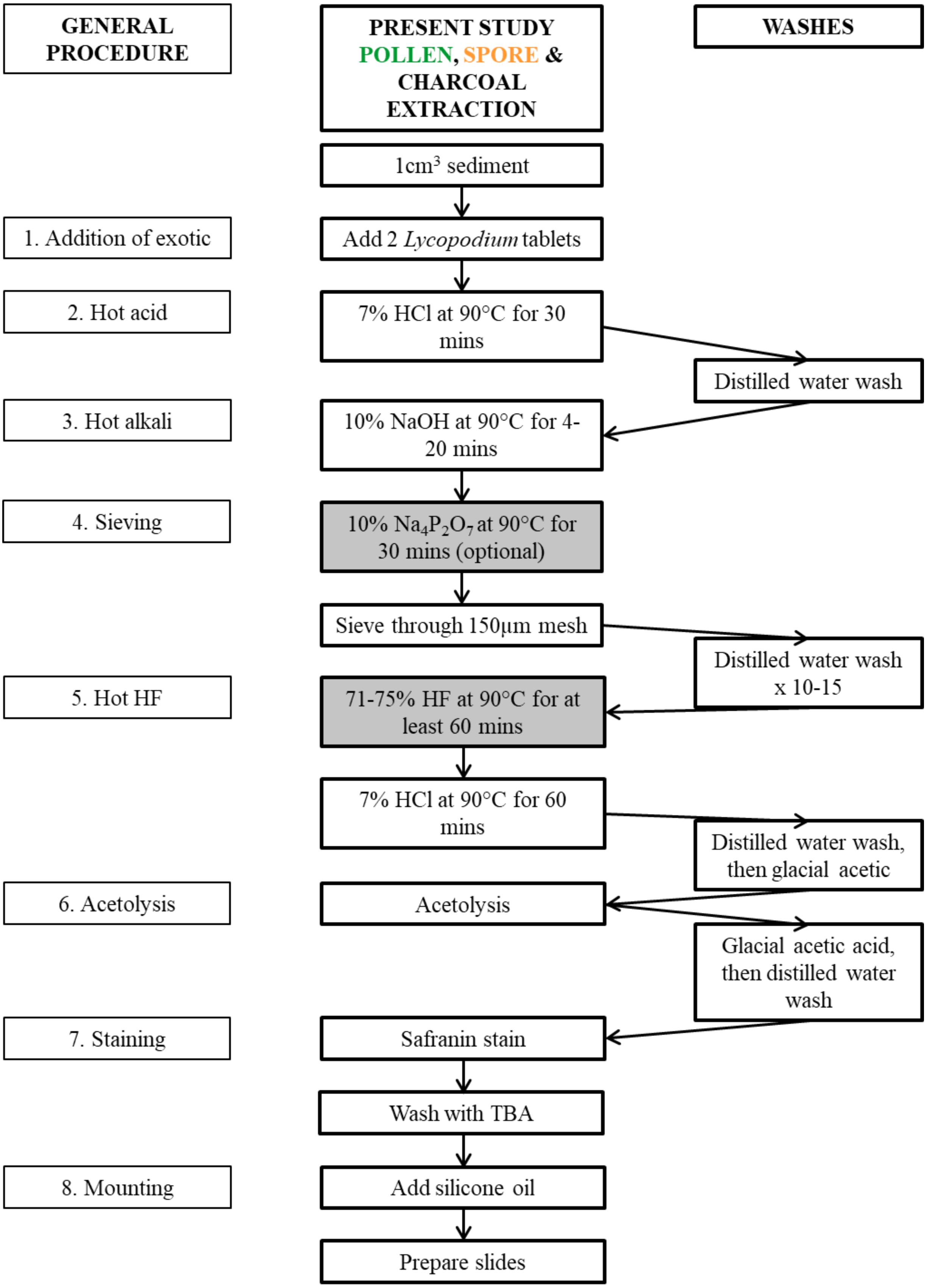



***Figure 4.1: Steps involved in the pollen, spore and charcoal extraction following standard general procedures as in Bennett and Willis (2002). Schematic adapted from Gillson (2002).***

#### *4.1.3.3 Diatoms extraction and analysis for RV3 core only*

Diatom analysis was initially and partially carried out by a UCT Honours student (Hoffenberg, 2019), however, the full dataset (increased number of sample depths and a minimum sum of 300) was recounted and analysed by a consultant (Kirsten, 2020) in order to provide a climate and hydrological proxy that was independent of the pollen data. Sediment sub-samples (1 $cm^3$) were collected at rangefinder depths from the GWWA006 and RV3 cores to test whether diatoms were preserved in the sediments. Only the RV3 core was suitable for diatom extraction. The same sub-sample depths as the other palaeoecological proxies were analysed for changes in diatom assemblages. Sub-samples were taken at 2-4 cm intervals along the 108 cm RV3 core and transferred to a 50 ml conical screw cap tubes. Diatom extraction was achieved by using standard methods to chemically treat the samples following a systematic process (Battarbee, 1986) summarised in Figure 4.2 with details described in Appendix 10.3.7. The general procedure sequentially removed extraneous materials such as organic matter (Step 1 and 3), minerogenic matter (Step 2 and 4) and fine clay particles (Step 5). Step 5 is crucial for achieving the main objective of obtaining samples that were satisfactory for reliable diatom identification and analysis. Prepared diatom samples (approximately 0.2 ml) were dry mounted using Pleurax (R.I. = 1.73) (Step 6, Figure 4.2) to allow for optimal visualisation of the diatom valves under the microscope.

Diatoms were identified using a Leica DM750 light microscope at 1000x total magnification, used in conjunction with immersion oil. To increase the visibility of the diatom sculpting, a green filter disc was added so that the light shining through the microscope was green in colour. For each level, a minimum sum of 300 diatom valves was counted. To avoid double counting of diatoms, fragments were only recorded if they included the valve centre or another single characteristic feature. Identification of diatom species was based on the Taylor et al. (2007) diatom guide for South Africa, and catalogues from various internet resources (e.g. https://diatoms.org/; http://www.algaebase.org/). Diatoms with significant autecology were grouped into categories which showed their preference for certain habitats and levels of salinity and nutrients. All diatom species were identified, however, when the preference of a species was unknown then the species abundance was summed and classified as ‘Unknown habitat’, ‘Unknown salinity’ and ‘Unknown nutrients level’. Certain diatom species were regarded as key indicator species that would provide insights on how the diatom assemblage from the RV3 palaeo-record changed over time, in response to drivers caused by climate and land-use change (Table 4.2).

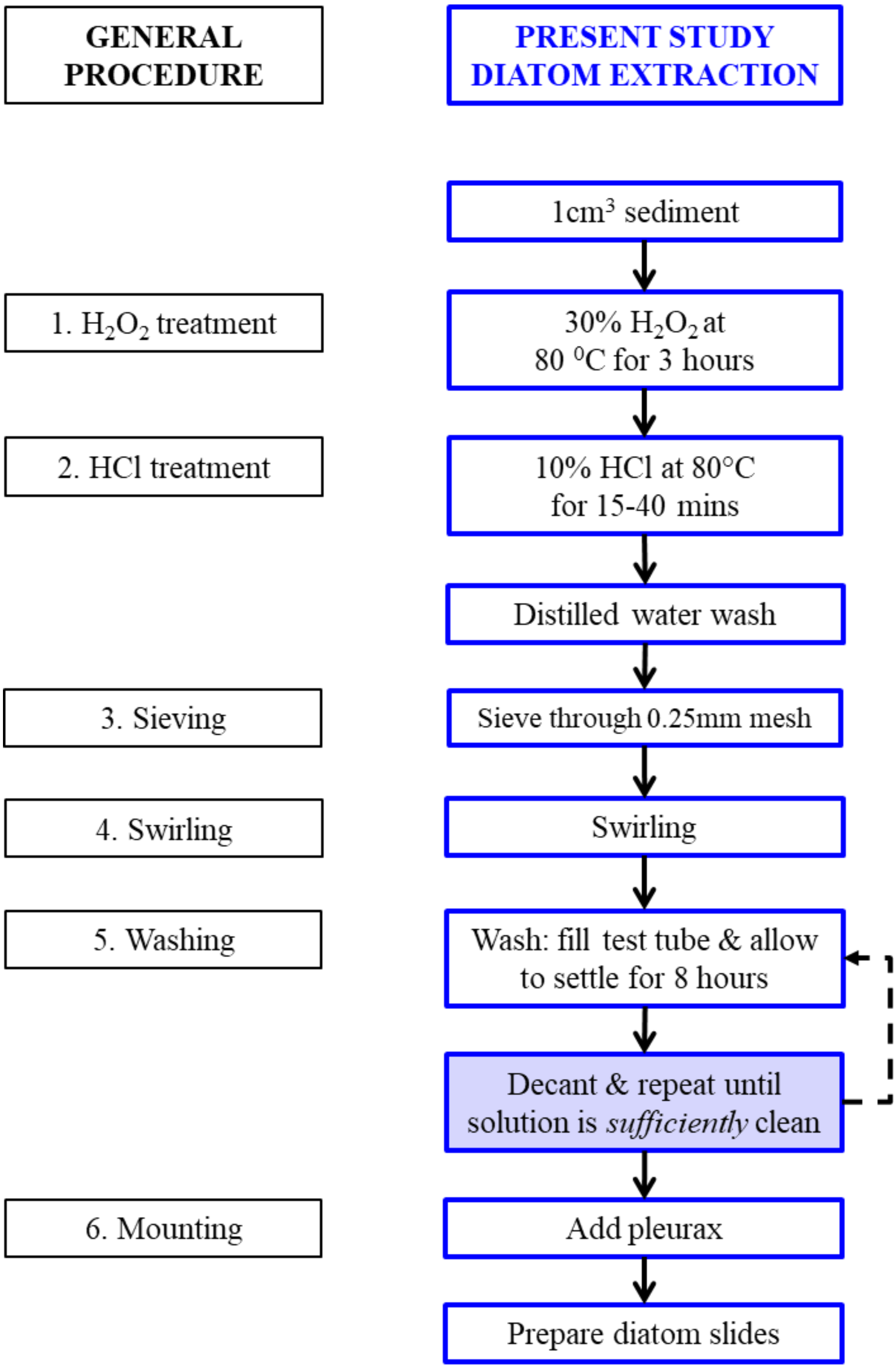



***Figure 4.2: Diatom preparation process adapted from Battarbee (1986) and Kirsten (2014)***

***Table 4.2: Diatom indicator species categorised according to autecology (habitat, salinity and nutrient preferences)*** (Gell et al., 2007; Kirsten, 2014; Stager et al., 2013, 2012; Taylor et al., 2007)**.**

| Preference | Description | Indicator diatom species |
|---|---|---|
| **Habitat** | Aerophylic (drying out) - Autochthonous | • *Hantzschia amphioxys*<br>• *Pinnularia borealis* |
| | Benthic (standing/open water) - Autochthonous | • *Staurosira elliptica* |
| | (Tyco)planktonic - Allochthonous. Note that tycoplanktonics and planktonics are not differentiated here as tycoplanktonics may have been transported but are likely autochtonous. Tycoplanktonics are easily detached from substrate so occur within planktonic assemblages (Wolin and Stone, 2010). | • *Aulacoseira granulata*<br>• *Melosira varians*<br>• *Cyclotella meneghiniana* |
| **Salinity** | Fresh | • *Aulacoseira granulata*<br>• *Achnanthidium minutissimum* |
| | Fresh-Brackish (low salinity) | • *Cyclotella meneghiniana* |
| | Brackish (medium salinity) | • *Nitzschia filiformis* |
| | Brackish-Saline (high salinity) | • *Gyrosigma acuminatum*<br>• *Tryblionella coarctata* |
| **Nutrients** | Oligotrophic (low nutrients including oligotrophic, dystrophic, and ultraoligo) | • *Achnanthes oblongella* |
| | Mesotrophic (medium nutrients including mesotrophic and meso-eutrophic) | • *Fragilaria biceps* |
| | Eutrophic (high nutrients including eutrophic, eu-hypereutrophic, eu-poly, and polytrophic) | • *Nitzschia filiformis*<br>• *Tryblionella levidensis* |

#### *4.1.3.4 Sediment description*

Sediment composition and particle size was assessed on waterlogged samples and described in terms of the abundance of four main components of sediment particles as devised by Troels-Smith (1955). The composition of minerals including clays and silt (*Argilla*) and fine and coarse sand and gravel (*Grana*) were determined according to the Troels-Smith (1955) protocols, thus sieving or a Mastersizer was not used to determine particle size. The description of plant components according to *Turfa*, *Detritus* and *Limus* (Table 4.3). The organic content was further assessed using Loss on Ignition (LoI) (see section 4.1.3.5.2 below). Furthermore, sediments within the cores were examined according to colour variation (Munsell, 1942)  and divided into units that had similar lithology. Changes in lithology stratification were recorded and plotted using the TILIA and TILIAGRAPH computer package programme (Grimm, 1997).

***Table 4.3: Troels-Smith (1955) classes used to describe the composition of sediment from cores retrieved at the upland conservation site (GWWA006) and lowland agricultural site (RV3).***

| Class | Element | Description |
|---|---|---|
| ***Turfa* (peat)** | Consists of macroscopic plant remains which are visible to the naked eye, and includes a term describing the humicity component. This is described using a five-point scale with 0 representing the absence of humic matter and 4 representing the complete dominance of humic matter. | |
| | *T. bryophitica* (Tb) | Consists of mosses. |
| | *T. lignosa* (Tl) | Consists of stump, roots, intertwined rootlets, trunks, stems, branches, etc., all lignified plant tissue. |
| | *T. herbacea* (Th) | Consists of roots, intertwined rootlets, rhizomes, stems, leaves, etc., all herbaceous plants. |
| ***Detritus*** | Consists of above ground parts of plants. | |
| | *D. lignosus* (Dl) | Consists of fragments of ligneous plants greater than 2 mm in size. |
| | *D. herbosus* (Dh) | Consists of fragments of herbaceous plants greater than 2 mm in size. |
| | *D. granosis* (Dg) | Consists of smaller fragments of ligneous and sometimes animal fossils, ranging from less than 2 mm to 0,1 mm in size. |
| ***Limus*** | Consist of aquatic mud made up from <0.1 mm fragments of plants and animals and includes a humicity component using the five-point scale described above. The small size of the *Limus* particles makes the subdivision of this class difficult. | |
| | *L. detrituosus* (Ld) | Plants and animals or fragments of these; particles <ca.0.1 mm ± humous substance (i.e., peaty). |
| | *L. ferrugineus* (Lf) | Has iron oxide present resulting in its red coloration and making it distinguishable from other *Limus*. |
| ***Argilla*** | Describes the mineral components of the sediments. | |
| | Clay (As) | <0.002 mm in size |
| | Silt (Ag) | 0.002 - 0.06 mm in size |
| ***Grana*** | Describes the mineral components of the sediments. | |
| | Fine sand (Ga) | 0.06 - 0.5 mm in size |
| | Coarse sand (Gs) | 0.5 – 2 mm in size |
| | Gravel (Gg) | >2 mm in size |

#### *4.1.3.5 Physical properties analysis (Magnetic Susceptibility and Loss on ignition)*

##### *4.1.3.5.1 Magnetic susceptibility (MS)*

Magnetic susceptibility (MS) is the degree to which a material can be magnetized in an external magnetic field. The measurement of MS, together with LoI as physical properties analyses, is as a palaeo-proxy for investigating changes in soil regulation processes and/or water content, and therefore categorised as a regulating service (see Table 4.4 in section 4.2.1 below). It is included in the study to specifically address Objectives 1 and 3 (Table 1.1). At the Department of Geography at Rhodes University, South Africa, a suite of magnetic measurements (a-c below) was initially carried out on a sub-sample of RV3 sediment depths using adapted procedures (Blumentritt and Lascu, 2015; Edwards and Rowntree, 1980; Foster et al., 2007, 1998; Magiera et al., 2006; Pulley et al., 2015). After receiving confirmation that MS measurements were feasible in both cores, volume magnetic susceptibility

(Volume MS in SI shown in (d) below) was measured at the Palaeoecology Laboratory in the Biological Sciences Department, UCT, using a Bartington *MS3 Magnetic Susceptibility Meter* (www.bartington.com). Further details on all magnetics procedures are found in Appendix 10.3.8.

(a) **Low-frequency magnetic susceptibility (**$\chi$lf in $10^{-6}$ $m^3$ $kg^{-1}$). High-frequency susceptibility ($\chi$hf), which is done for samples with low-frequency values of greater than 20 $\chi$lf.

(b) **Anhysteretic remanent magnetisation (ARM):** ARM was carried out at 40 micro-Tesla.

(c) **Isothermal remanence (IRM):** IRM was measured at 0.8 Tesla with magnetisation gradually increasing from 20 milli-Tesla to a maximum of 1 Tesla (saturation IRM or SIRM) to generate the IRM curves. After each increase in, remanence was measured. After 24 hours, viscous loss of remanence was measured, which involved measuring IRM again. Back IRM was measured at -0.1 Tesla.

(d) **Volume magnetic susceptibility (Volume MS) (unit SI):** MS is a dimensionless quantity measured in units expressed as SI or CGS. Note that in the CGS system the unit of length, mass and time are centimetre (cm), gram (g) and second (s), respectively. Whereas in the SI system the unit of length, mass and time are metre (m), kilogram (kg) and second (s), respectively.

#### *4.1.3.5.2 Loss on Ignition (LoI)*

Loss on Ignition (LoI) estimated the organic carbon content ($LoI_{550}$), inorganic carbon/carbonate content ($LoI_{1000}$) and water content of sediments (Bennett and Willis, 2002). The protocol of Heiri et al (2001) was followed for the analysis of LoI. By using a 3 ml syringe with the tip removed, 1 $cm^3$ samples were removed from the core at the same resolution as for other proxy analysis at between 2-4 cm intervals throughout the core. Samples were sequentially heated in a muffle furnace, recording five sets of weights were recorded followed by a series of calculations (A-E) detailed below (Bengtsson and Enell, 1986; Walter E. Dean, 1974).

A. A labelled set of dry crucibles was weighed and recorded (A).
B. A known volume of sediment was transferred to each crucible and a new weight (wet sample + crucible) was recorded (B).
C. Samples were dried overnight for 16 hours to constant weight at a temperature of around 100°C and the dry weight recorded (C).
D. Samples were exposed to a temperature of 550 °C ($LoI_{550}$ or the carbon content) for five hours, to remove organic carbon (organic matter oxidised into carbon dioxide and ash). Cold samples

were put into the muffle furnace, the timer was only set once the furnace had reached the desired temperature (in this case 40 minutes after the sample had been put in). Samples were allowed 5 hours to cool whilst in the furnace. Crucibles were placed in a desiccator when they were removed from the oven and between weighing to prevent absorption of moisture from the air, which would affect weights (D).

E. In the final step samples were heated at 1000 °C ($LoI_{1000}$ or the carbonate content) for two and a half hours, enabling carbonate content to be calculated from the final weight of the sample (Walter E. Dean, 1974). The oven was then switched off and the samples left to cool overnight (12 hours) to be reweighed in the morning (E).

The following equations adapted from Heiri et al (2001) and Gillson (2002) were used where the symbols A-E refer to the steps outlined above:

$$Percentage\ water\ (\%) = \frac{(B-A)-(C-A)}{B-A}$$

$$Dry\ Weight\ (DW) = C - A$$

$$Percentage\ organic\ carbon\ (\%\ \mathrm{LoI}_{550}) = \left(\frac{(C-A)-(D-A)}{C-A}\right) * 100$$

$$Percentage\ carbonate\ content(\%\ \mathrm{LoI}_{1000}) = \left(\frac{(D-A)-(E-A)}{C-A}\right) * 100$$

#### ***4.1.3.6 Geochemical analysis of sediments***

Elemental composition of sediments was measured using an XEPOS XRF (SPECTRO Analytical Instruments GmbH, Germany) (Figure 4.3). Sediment sub-samples from the GWWA006 and RV3 cores (5 g wet wight) were dried over night for 16 hours to constant weight at 100°C. Samples were ground to a fine powder using a mortar and pestle or a ball mill (MM200, Retsch, Germany) (Step 1). The samples were placed in sample cups that were Perspex cylinders sealed with 4-µm Polypropylene Thin Film (Chemplex Industries Inc, Florida, USA) (Step 2).

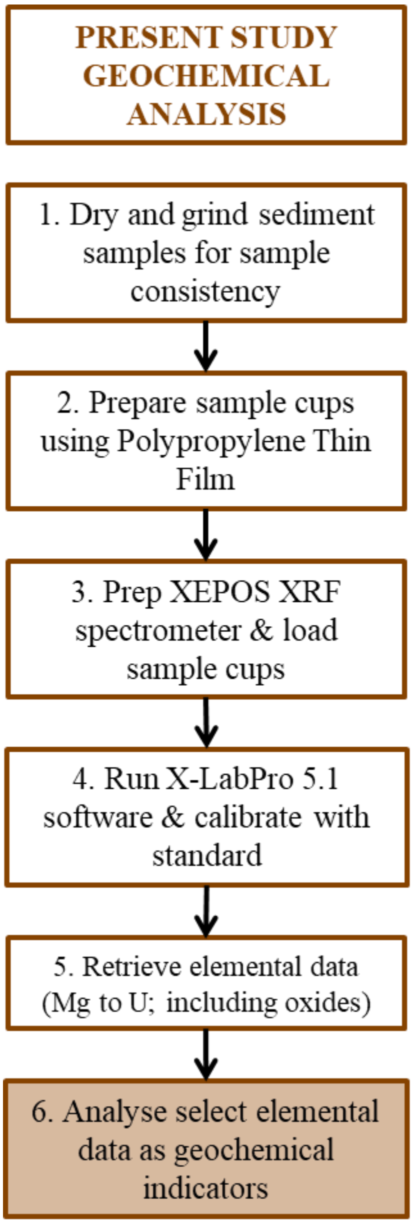



***Figure 4.3: Summary of procedure followed to determine elemental composition of sediments.***

Sample cups were loaded into a SPECTRO XEPOS XRF spectrometer to measure the geochemistry of the sediments (Step 3). Analyses were conducted using the X-LabPro 5.1 software, which incorporates the universal 'Turbo Quant Powders' methodology (Step 4). The XRF spectrometer was calibrated by using a certified GSS1 geochemical reference standard. Elemental concentrations (mg $kg^{-1}$ expressed as % in the geochemical esults) of elements that were within the XRF spectrometer's detection limits were obtained (Mg to U on the periodic table) (Step 5). However, only several elements were selected as suitable geochemical indicators for soil regulation (Step 6): Magnesium (Mg); Phosphorus (P); Potassium (K); Calcium (Ca); Manganese (Mn) and Iron (Fe) (trace metals);

Zinc (Zn) and Copper (Cu) (heavy metals); Caesium (Cs), and Barium (Ba) (algal blooms) (Baioumy et al., 2010; Collins et al., 1997; Dearing et al., 2012; Dymond et al., 1992; Geel, 2020; McGregor, 2014; Prakash Babu et al., 2002; Sadeghi et al., 2014; Virah-Sawmy et al., 2009). Furthermore, analysis of ratios of select elements were used to interpret the geochemical data together with other palaeoenvironmental proxies that show changes in vegetation, rainfall, herbivory.: Zirconium to Rubidium ratio (Zr:Rb) (erosion and aeolian activity) (Liu et al., 2004; Rydberg et al., 2016); and Rubidium to Potassium ratio (Rb:K) (palaeo-salinity ratio since $Rb^{+}$ concentrations tend to be higher in marine water (0.12 ppm) than in fresh water (0.0013 ppm) (Dasch, 1969; Rydberg et al., 2016; Taylor and McLennan, 1985; Yang et al., 2020).

### 4.1.4 Exploratory statistical analysis and software

Methodological flowcharts and various summary figures of results were created in Microsoft® Office PowerPoint 2010 edition. GIS (geographic information system) maps were constructed using Quantum GIS (QGIS) Version 3.12 (Development Team, 2009, QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org).

#### *4.1.4.1 Changes in palaeoenvironmental proxies over time*

Although the analysis of quantifiable palaeoenvironmental data largely describes patterns and trends over time, other quantitative statistical techniques are used to enhance the classification, description and therefore interpretation of complex multivariate stratigraphic data (Birks and Gordon, 1989; Blaauw et al., 2019; Seddon et al., 2014). Pollen, spore, charcoal, diatoms, physical properties (MS and LoI) and geochemical data were initially prepared in Microsoft® Office Excel 2010 edition and descriptive statistical analyses performed in R. Before carrying out further data analysis such as the zonation and ordinations, data were normalised by applying the square root transformation to enhance the representation of types with low abundance (Birks et al., 2012). All the palaeoenvironmental proxy data were plotted using C2 version 1.7.7 (Juggins, 2007), a Microsoft Windows programme for developing and applying palaeoecological transfer functions and for visualising multi-proxy stratigraphic datasets. An online manual was used for guidance, see https://www.staff.ncl.ac.uk/stephen.juggins/software/code/C2.pdf.

Fossil pollen and spore data were described in both percentages and concentrations to facilitate comparisons and the identification of patterns that are robust across both data types, and therefore confidence in the general trends seen in the proxy data. If there were differences between the percentage and concentration results the reason is due to the Fagerlind effect in the percentage data (Fagerlind, 1952).

Quantitative analytical techniques such as zonation (cluster analysis), allow the demarcation of statistically significant portions of the palaeo-stratigraphy (Bennett, 1996; Birks and Gordon, 1989; Jackson, 1993), and have been used to guide the interpretation for potential alternative stable states or regime shifts that are ecologically and environmentally significant (Forbes et al., 2018; Gil-Romera et al., 2010; MacPherson et al., 2018, 2019; Quick et al., 2015). One such technique uses the 'constrained hierarchical clustering' method, which identifies the number of statistically significant clusters in pollen percentage and concentration data as well diatom percentage data. The steps below were applied to the pollen and diatom data separately to detect whether changes in multi-proxy data were synchronous or not. To do this, packages '*rioja'* (Juggins, 2015) and '*vegan'* (version n 0.9-15) (Oksanen et al., 2013) were used in R Studio (R Studio Team, 2015) (see cluster analysis details and R code in Appendix 10.3.10.1).

#### *4.1.4.2 Comparing palaeoenvironmental proxies*

In contrast to the zonation technique, in a non-metric multidimensional scaling (NMDS) ordination, data are not stratigraphically constrained and can reveal ecological relationships between temporally distal types (pollen species and other proxies) and samples (depths) (Legendre and Birks, 2012; McCune et al., 2002). The analysis was performed in RStudio using the *Vegan* package (Oksanen et al., 2013) and was performed using the pollen percentage data only (see details in Appendix 10.3.10.2).

## 4.2 Boundary crossing approach to apply the past-present-future lens of environmental change

### 4.2.1 Supporting Components A and B to identify drivers of change (Objective 2) and interpret data using ecosystem services (Objective 3)

The identification of drivers of change (Objective 2) and interpretation of palaeo-data in terms of ecosystem services and resilience (Objective 3) were enhanced by the inclusion of supporting methodological inputs, Component A and B (Figure 1.2 and Table 1.1). Quantifying ecosystem services indices for supporting, provisioning and regulating services or an overall ecosystem service index is not within the scope of this study. However, being explicit about the effort to find a "common language" to translate the palaeo-data into meaningful metrices for land-use management and policy

(Table 4.4) is an important element to bridge the gap between reconstructing environmental history and developing a SD model:

- **Component A:** Literature was reviewed to identify the array of palaeoenvironmental proxies that are suitable to investigate changes in select ecosystem services and drivers over time. Relevant palaeoecological and historical literature helped draw up historical timelines on specific events related to direct and indirect land-use impacts and climate change in the region (see Table 5.1 and Table 6.1 later).
- **Component B:** Stakeholder engagement determined the links and causal relationships between select ecosystem services, drivers and stakeholders in the region. Such expert input contextualized the environmental and socio-economic SES problems that are within and outside stakeholders' ambit. Stakeholder engagement, within participatory system dynamics (PSD), included key expert input important for variable elicitation for model development (see section 4.2.3.1 below).

***Table 4.4: Selected list of palaeoenvironmental proxies associated with a description of the ecosystem services (ES) category and the ES processes or states they represent.***

| Ecosystem services (ES) category | ES process/state | Palaeo-proxy/indicator (Type of analysis) | Type of information obtained | Reference |
|---|---|---|---|---|
| Provisioning services (genetic resources) or Supporting services (primary production - biomass) | Plant biodiversity | Pollen analysis (section 10.4.3) | Past vegetation reconstruction based on changes in fossil pollen assemblage. | (Bennett and Willis, 2002; Gil-Romera et al., 2010; Gillson et al., 2020; Quick et al., 2011; Scott and Woodborne, 2007a, 2007b) |
| Regulating services *Driver of change* related to climate regulation | Water quality and climate change | Diatom analysis (section 4.1.3.3) | Past habitat, salinity, and nutrient flux reconstruction to climatic and land-use changes. | (Bennion et al., 2011; Gell, 2012; Gell et al., 2005; Kirsten, 2014; Stager et al., 2012) |
| Regulating services | Soil regulation | Sediment description (Troels-Smith and Munsell soil colour) (section 4.1.3.4) | Composition and colour changes of the sediment core. | (Munsell, 1942; Troels-Smith, 1955) |
| Regulating services | Soil regulation (and water content) | Physical properties analysis (Magnetics and Loss on ignition) (section 4.1.3.5) | Estimation of erosion rates (sediment source), water, organic carbon, and carbonate content of sediments | (Bengtsson and Enell, 1986; Bennett and Willis, 2002; Dean Jr, 1974; Heiri et al., 2001) |
| Regulating services | Soil regulation | Geochemical - elemental composition (section 4.1.3.6) | To investigate soil erosion regulation (sediment source) and water cycling, particularly the source of sediments. | (Baioumy et al., 2010; Collins et al., 1997; Dearing et al., 2012; Geel, 2020; Sadeghi et al., 2014; Virah-Sawmy et al., 2009) |
| *Driver of change* related to Provisioning services (meat/wool/diary) | Herbivory (land-use disturbance) | Spore analysis (section 10.4.4) | Past herbivory/grazing reconstruction based on changes in coprophilous fungal spore assemblages. | (Davis and Shafer, 2006; Ekblom pers. comm., 2013; Gelorini et al., 2012; Graf and Chmura, 2006) |
| *Driver of change* related to Provisioning services (land clearing/stimulation of grass growth for pasture) | Fire (land-use disturbance) | Macro-charcoal (>150 µm) and micro-charcoal (>10 µm) (section 10.4.5) | Local and regional fire history reconstruction based on changes in macro- and micro-charcoal abundance, respectively. | (Bennett and Willis, 2002; Clark, 1982; Duffin et al., 2008; Mooney and Tinner, 2011) |

### 4.2.2 Framing the past-present-future lens into Storylines and Scenarios

An adaptation of applied thematic analysis was used to synthesise the insights from the integration of qualitative and quantitative mixed methods (Process Steps 1-9, Table 4.5) (Guest et al., 2012; Namey et al., 2007). The conceptual meta-framework underpinned the inputs and layout of the thematic narratives. Results were collated into Storylines (*past* and *present*) and Scenarios (*future*) that could be explored as a problem statement during follow up engagement with interested stakeholders. Storylines included: **Patterns of the past** derived from long-term palaeo-data from Elandsberg PNR to identify drivers and interpret ecosystem services and resilience to develop a picture of the past SES problem. **Narratives of the present** derived from stakeholders' narratives of change (i.e. their beliefs, values and assumptions as mental models) highlighted using key quotes. The general narrative of perceptions was captured in a summarised CLD version of the systems map. **Scenarios for the future** were hypothetical scenarios developed to explore possible futures of how and why the SESs will change in the future under various environmental and socio-economic influencing factors.

### 4.2.3 Participatory System Dynamics (PSD) Modelling Approach (Objective 4, Components B and C)

System dynamics (SD) is a rigorous way to help us think, visualise, share and communicate complex systems and issues over time. SD, as a technique to understand change and unravel complexity, focuses on modelling the problem so that we can create more robust designs which minimise the likelihood of unpleasant surprises and unintended negative consequences. Figure 4.4 shows the generic SD modelling process recommended by Sterman (2000) which includes five essential iterative phases: Phase 1 and 2 include the qualitative components and Phases 3-5 include the quantitative components.

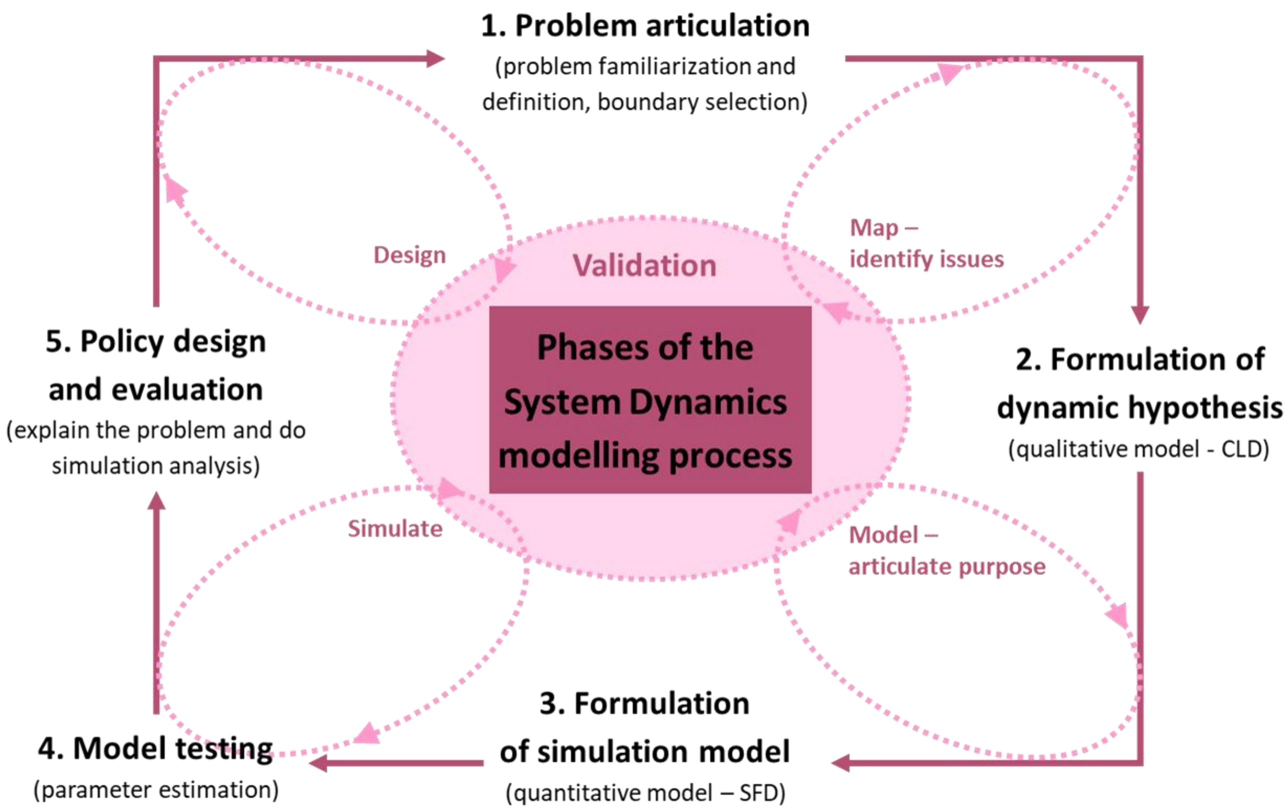



***Figure 4.4: Schematic showing the iterative nature of the five phases of the System Dynamics modelling process (maroon solid line) and an iterative model validation approach (light pink dashed line) for quality assurance. Adapted after Schwaninger and Groesser, 2009; Sterman, 2000.***

In the present study, all five phases of the SD process (Figure 4.4) was followed in addition to mediated modelling principles (van den Belt, 2004). Therefore, the approach followed is regarded as participatory system dynamics (PSD). In addition to capturing the structure and connectedness of complex SESs, PSD regards the perception of the stakeholders as an integral part of model development. Furthermore, similarly to 'Agile System Dynamics' methodologies, PSD assumes that the project objectives (Figure 1.2) are not addressed in isolation from each other and broad principles such as revision, reflection, learning and technical excellence is encouraged, thus providing increased confidence in the overall model results (Warren, 2014).

#### *4.2.3.1 Research Process Steps and Elandsberg PNR case study as a proof of concept*

Noteworthy process steps to achieve the overall research aim of this study, which is to provide a past-present-future lens of environmental change in the study area to contribute to sustainable management of ecosystem services are described in Table 4.5. Although Process Steps 1, 2 and 4 are not a part of typical SD research project, Table 4.5 shows how the palaeoecological-related objectives (Objectives 1-3, Component A) and PSD modelling-related objectives (Objectives 4 and Components B and C - Figure 1.2) are linked to the fives phases of the SD modelling process.

Original palaeoecological data and modern vegetation data from GWWA and RV3 was collected (Process Step 1 and 4, Table 4.5). Insights from grey and academic literature on social-ecological processes and long-term climate and land-use history within in the CFR was captured (Process Step 2, Table 4.5). Initial preparation and stakeholder interactions were facilitated via introductory semi-structured interviews with commercial farmers, conservation practitioners and government authorities on different occasions between 2016-2021 to inform the conceptual ideas and the development of an initial CLD. The semi-structured interviews allowed for the gathering of qualitative data since stakeholders were given the freedom to reflect, brainstorm ideas and tell stories (Process Step 3, Table 4.5). However, these interviews were considered as perspectives, and therefore ‘clues’ for the use of other more suitable sampling/data collection strategies in the subsequent process steps. During the study period, certain stakeholders emerged as ‘key informants’ based on their formal roles in an organisation. This means that key informants provided generalized information about patterns of behaviour, after summarising either observed (actual) or expected (prescribed) organisational operations (Clifford-Holmes, 2015; Kumar et al., 1993). During semi-structured interviews a generic template (Appendix 10.4.2) was used to capture the perspectives. The sampling strategy of semi-structured interviews is shown in Appendix Table 10.1, including information on several key informants from different backgrounds and organisational landscapes (Voss et al., 2002). The ethics clearance certificate and consent form are included in the Appendix Figure 10.3 and Figure 10.4, respectively.

Leading up to the multi-stakeholder engagement workshop a rapport was established with stakeholders via email correspondence and informal meetings/semi-structured interviews. The multi-stakeholder engagement workshop was held at the LandCare offices in Wellington, Western Cape, South Africa on 4 July 2019. Participants were invited because of their involvement with land-use management in the Middle Berg River Catchment, or because of their knowledge of EbA, conservation and agriculture within the region. Since project conception in August 2016, key stakeholders were identified when selecting the case study sites where sediment cores were located (GWWA, Rhenostervlei Farm and Elandsberg PNR) and by my participation in various EbA workshops.

The workshop was facilitated with several key informants present to co-create understanding in a mediated and participatory manner (Process Step 5, Table 4.5). Some of the ethical considerations in the present study included transparency about the research aims and procedures, and the limitations thereof. As the lead facilitator, I was experienced as a Table Host using this social learning process in two prior workshops, but I had not led such a social learning process with multiple stakeholders within the present study before. However, I was experienced in co-facilitating workshops with small to large groups of various stakeholders (farmers, community members, school learners) covering topics including, general project planning and environmental Monitoring and Evaluation (M&E); Earth System Science; and Environmental and Social Safeguards. I planned, organised and facilitated the multi-stakeholder engagement workshop. Planning included training of a facilitation team: one expert SD modeller; one scribe; and three Table Hosts (one research assistant from the Plant Conservation Unit and two post-doctoral researchers in SESs). Table Hosts play an essential role in the multi-stakeholder workshop as they were required to be impartial and neutral, while mediating the dialogue to encourage fair contribution amongst participants and capturing feedback in a transparent way (see page 3/6 in Appendix Figure 10.2). Furthermore, ethical considerations about anonymity were communicated as written notes, audio recordings and photographs were captured during interactions with stakeholders, and they were reassured of their anonymity for reporting and publication purposes (Appendix Figure 10.4).

Although the World Café and Sustainability Dialogues methodologies are not explicitly classified as social learning methodologies, their approach encourages reflection (Anderson, 2011) and co-creation (congruent with collective innovation - Hechenbleikner et al., 2008; Oelofse and Cady, 2012). Furthermore, Currie (2018) applied a similar methodology for stakeholder engagement workshops to improve public participation in water management and environmental governance within the Knysna Municipality, Garden Route, South Africa. The multi-stakeholder engagement workshop approach (Table 10.2) was therefore seen as a suitable process step to enhance insights and achieve Objectives 2, 3 and 4. The multi-stakeholder engagement workshop consisted of the following aspects (see Appendix Table 10.2 and Figure 10.2 for further details):

- **Setting the scene**: I gave an initial presentation on the project purpose, and objectives and process of the multi-stakeholder engagement workshop. Setting the scene included creating a hospitable environment for participation (a relaxed “coffee shop” setting). Scientific jargon and terms such as EbA, ecosystem services (communicated to stakeholders as “natural resources”) and drivers of change including land-use disturbance and climate change (communicated to stakeholders as “influencing factors”) were discussed to establish a ‘common language’ and harmonize vocabulary or jargon essential for meaningful knowledge exchange and co-creation (Biggs et al., 2008; Marcisz et al., 2018).

- **Dialogues**: These included facilitated discussions around natural resources (Ecosystem Services - Dialogue 1), influencing factors (concerns/threats/drivers of change - Dialogue 2), and connections (links/relationships/cause and effect - Dialogue 3). Each dialogue had two or three iterations so that stakeholders could discuss the same topic with different stakeholders during each iteration. Materials included large pieces of paper, sticky notes, index cards, pens, a talking jar and the ‘parking lot’ on brown paper.
- **Feedback**: After each dialogue break-out session, the participants came together for feedback during plenary before moving onto the next dialogue topic.

Process Steps 3 and 5 were an integral part of the model building process as it involved consultation with stakeholders for problem elicitation (see Figure 4.4 above and Table 4.5 below) to gain a holistic systems perspective of SESs within the Middle Berg River Catchment, including the role of stakeholders and their perceptions of changes in the ecosystem services and drivers. Further active modes of stakeholder contribution such as the co-construction of models’ structures (see Process Steps 9 and 10 below, Table 4.5) were outside the scope of the present study. The rationale for this was related to the project aim and objectives to use palaeoecology and stakeholder engagement for the development of a SD model. The simulation model was intended as a land management decision-support tool as opposed to a decision-making tool, which would require an even higher level of engagement and input from key stakeholders.

However, despite no further co-development with stakeholders, the qualitative data generated from both the multi-stakeholder workshop and follow-up interviews with farmers were used in subsequent steps of the model building process. Qualitative data collected during the multi-stakeholder engagement workshop was analysed thematically using the Iceberg Model (Process Step 6, Table 4.5) to gain a multi-level systems understanding of sustainable land-use management (see section 2.3, Figure 2.3). Moreover, causal links between variables as identified by stakeholders was represented diagrammatically in a Systems Map (Process Steps 6 and 7, Table 4.5).Via such causal diagrams developed by collective learning by land-use managers, qualitative data was integrated to provide a systemic snapshot of how social and biophysical drivers in the context-specific Middle Berg River Catchment are interlinked. Thus, revealing multiple processes that operate simultaneously to cause and exacerbate SES problems such as land degradation or the loss of ecosystem function. During the workshop stakeholders also shared insights on changes in plant biodiversity in response to fire and grazing management over time. These insights were later used to produce future scenarios during model analysis (Phase 5 in Figure 4.4 and Process Step 11 in Table 4.5), which allowed for the exploration of changes in ecosystem services under various land-use and climate conditions that influence the SES behaviour towards a sustainable state.

***Table 4.5: Process steps and tools used in the present study. Each step is related to the five phases of the system dynamics (SD) modelling process and the Objectives and Components. Steps 8-13 marked with an asterisk(*) apply only to the Elandsberg PNR case study. Stakeholder contribution to the model building process is shaded grey.***

| Phases of SD modelling process (Sterman 2000) | Objective /Component in this study | Tool used | Process step | Description of process step used in this interdisciplinary study |
|---|---|---|---|---|
| Phase 1 | Objective 1-3 | Multiple palaeo-environmental proxies | Step 1 | Environmental history was reconstructed using multi-proxy, high resolution palaeo-proxies over decadal-centennial-millennial timescales to illustrate the time series and identify troubling trends that were unprecedented. Extraction and analysis of pre-selected palaeo-proxies from sediment cores included: pollen, spores, charcoal, diatoms, sediment description (Troels-Smith and Munsell colour) and physical properties (magnetics and LoI) and geochemical (elemental concentration - XRF). Exploratory statistical analyses (cluster analyses, NMDS) and calculus provided understanding of ecological dynamics. |
| | Component A | Historical timelines: Using palaeo and historical literature | Step 2 | To establish the socio-ecological context, historical timelines were drawn up using relevant palaeoecological and historical literature on specific events related to land-use and climate change in the region. Since hypotheses of the trends represent our surface mental models (beliefs, values and assumptions), the palaeo-data was important to identify emergent patterns (i.e. prevalence of the behaviour equates to the prevalence of the outcome) and the historical timelines related the patterns to specific events. |
| | Component B | Semi-structured interviews | Step 3 | Semi-structured interviews were conducted with stakeholders from the three study sites to gain insights regarding the SES context to assist with palaeo-data interpretation (GWWA and RV3) and model development (Elandsberg PNR). Appendix 10.4.2. |
| | Objective 1-3 | Vegetation surveys | Step 4 | Vegetation surveys were conducted to understand the landscape dynamics at the study sites and aided in interpreting the palaeo-data. Appendix 10.2. |
| | Component B | Multi-stakeholder engagement workshop | Step 5 | An adapted GMB script and social learning process methodology was used for variable elicitation during a multi-stakeholder workshop with key informants. Model elements (variables) were identified via dialogues that encouraged reflection and co-creation. Appendix 10.4.3. |
| | Component B and Objective 4 | Iceberg model | Step 6 | Recorded notes from the multi-stakeholder engagement workshop were collated by Table Hosts and used in a thematic analysis by applying the Iceberg Model – a tool that illustrates a systems perspective (Monat and Gannon 2015). Variables were clustered according to the four levels (Events/Consequences; Patterns/Behaviours; Systemic Structures; and Mental Models/Mindsets/Physical or Chemical Forces). The analysis was done separately for ecosystem services (Dialogue 1) and Drivers (Dialogue 2). |

| Phases of SD modelling process (Sterman 2000) | Objective /Component in this study | Tool used | Process step | Description of process step used in this interdisciplinary study |
|---|---|---|---|---|
| Phase 1 | Objective 4 | Systems mapping | Step 7 | Connections between elements/variables elicited during the multi-stakeholder engagement workshop (Dialogue 3) were recorded using modelling software (Stella Architect). One-way arrows were used to connect the influence and effect variables, thus indicated whether they increase or decrease in size or quantity. This resulted in a systems map – Appendix Figure 10.24. |
| | Objective 1-3 | Reference modes | *Step 8 | Elandsberg PNR case study was chosen as a proof of concept for applying a past-present-future lens through which sustainable land-use management can be viewed. Fossil pollen, coprophilous fungal spores and macro-charcoal data spanning ca.1300 years (Forbes et al. 2018) provided a reference mode for Causal Loop Diagram (CLD) and Stock and Flow Diagram (SFD) development. |
| Phase 2 | Objective 4 | Qualitative model development (CLD) | *Step 9 | Based on the palaeo-data (time-series as reference modes) and systems map generated from outputs of the stakeholder engagement, an appropriate model boundary was identified for the Elandsberg PNR SES problem. A CLD (a dynamic hypothesis) was developed to illustrate the problematic behaviour (using modelling software Stella ® Architect, 2019. isee systems inc). |
| Phase 3 and 4 | Objective 4 | Quantitative model development (SFD) | *Step 10 | A SFD was developed based on the CLD, using insights from the palaeo-data, literature, and expert input. Formulation and parameter estimation was completed. |
| Phase 5 | Objective 4 | Model analysis (scenarios and policies) | *Step 11 | Policies linked to recommended fire and grazing management thresholds were designed and evaluated via simulation analyses (future scenarios of changes in plant biodiversity in relation to changes in fire and grazing) and sensitivity analysis. |
| Phase 1 | Component C | Management decision-support tool | *Step 12 | The end-product included an interface (Story Interface in Stella ® Architect, 2019.) to help Elandsberg PNR reserve managers set sustainable land-use guidelines/strategies on future grazing and fire that would protect local plant biodiversity, and safeguard ecosystem services and build resilience in the region. The interface allows for engagement with interactive data visualisations powered by the SFD and can be used as a boundary object to facilitate dialogue about any surprising simulation results. |
| | Component C | Past-present-future lens of environmental change: Scenario-planning | *Step 13 | Recommendations were made about resilience and restoration. These were based on the desire to maintain local plant biodiversity based on policy sensitivity analysis of modelled results with pre-1950s levels of grazing and fire. Recommendations provided information for the exploration of future scenario-planning at a follow up workshop with interested stakeholders as the social learning process originally intended. |

Process Steps 1-7 applied to land-use management at all three case study sites within the Middle Berg River Catchment in general. However, given the scope and the integration of paleoecology and PSD in the present study, subsequent Process Steps 8-13 (see asterisk (*) in Table 4.5) were only based on previous published findings (Elandsberg PNR) and not on the new palaeoecological datasets from GWWA and RV3. The knowledge generated in previous steps helped gain a holistic view of the SES problem and set an appropriate boundary for the modelling problem explored via the Elandsberg PNR case study (Process Step 8-13, Table 4.5). Therefore, the Elandsberg PNR case study could be used as a 'proof of concept' for blending palaeoecology and PSD into management guidance. A unique contribution to the initial phases of the model building process was the use of fossil pollen (plant biodiversity proxy), coprophilous fungal spores (herbivory proxy) and macro-charcoal (local fire proxy) data spanning ca.1300 years (Forbes et al. 2018) as reference modes (Process Step 8, Table 4.5).

Selected statistical relationships from the VANG core palaeo-data derived from NMDS, calculus and correlation analysis between plant biodiversity, herbivory and fire provided insights that were used to augment causal relationships illustrated by a causal loop diagram (CLD) developed using modelling software called Stella (Stella ® Architect, 2019. isee systems inc) (Process Step 9). CLDs allowed for identifying interrelationships among a set of variables operating in a system and patterns of change as opposed to static snapshots (Senge, 1990). The basic elements of CLDs are variables and arrows (links/connections). A variable is a condition, situation, action or decision that can influence and be influenced by other variables. A variable can be a qualitative or soft variable, for example, motivation, trust, buy-in, willingness, or quantitative and measurable such as agricultural land, food production, vegetation cover. An arrow indicates a causal association between two variables or a change in a state of these variables. The causal relationship can either happen in the same direction ("s" or "+" or positive correlation) as the arrow/link or it can happen in the opposite direction ("o" or a "-" or negative correlation). As the direction of the relationship between two variables is not fixed and can change under different assumptions, circumstances or within a different range of variables, the systems map and CLDs were used to develop a theory or preliminary conceptual and qualitative model about how the connections in the system give rise to the SES's behaviour and patterns seen in palaeo-time series data. The CLD described the dynamic hypothesis of the ecological problem based on the two interacting sectors: ecosystem services and drivers of change. Causal-thinking is not necessarily correlational but rather that causal or feedback loops can either be reinforcing or balancing/counteracting loops. Changes in the connections may allude to changes in the systemic structure of the SES (Bou Nassar et al., 2020; Inam et al., 2015; Langsdale, 2007; Videira et al., 2010; Voinov and Bousquet, 2010).

Both dynamic- and causal-thinking was represented by stocks and flows with quantitative accumulations. Therefore, the way in which these variables (i.e. stocks (levels), flows (rates) and converters (auxiliaries)) represent a system is critical to the dynamics of the system. Based on this preliminary CLD, the stock and flow diagram (SFD) (Process Step 10, Table 4.5) was developed by representing the ecological system as a set of stocks (Renosterbos pollen, Asteraceae pollen, macro-charcoal and coprophilous fungal spores) and flows (e.g. increase and decrease in plant diversity, fire and grazing in and out of the stocks). Given the model boundary, a number of variables were excluded from the stock and flow diagram (SFD) presented in section 7.2.2, such as micro-charcoal as a proxy for regional fires, exotic pollen as a proxy for alien invasive plants, area (ha) under protection/stewardship, "old lands" area (ha) previously under cultivation, and climate parameters such as rainfall and temperature. Further details about the SFD formulation, model testing and analysis are described in Chapter 7.

Process Step 11 and 12 (phase five of the SD modelling process in Figure 4.4 and Table 4.5) comprised of the evaluation of policies. Based on the palaeoecological historical range of variability, potential management thresholds were tested without the calibration and direct quantification of fossil pollen, coprophilous fungal spores and macro-charcoal to vegetation cover, large stock units and fire intensity. However, despite the lack of calibration, this simulation experiment allowed for the exploratory analysis of future scenarios which included possible combinations of grazing and burning at the Elandsberg PNR site (see section 7.3). Therefore, the systems map, qualitative CLDs, and quantitative SFD are intended as support tools for decision-makers since it fulfils the present study's case-specific purpose: *To help Elandsberg PNR managers set land-use guidelines/strategies on future grazing and fire that would protect local plant biodiversity and therefore ecosystem function.* A decision-making tool is not within the scope of the present study, however, I show how research findings can inform policy implications and management by including end-products in a 'toolkit' generated from the mediated process. The end-products included CLDs, a systems map, Stories and Scenarios for Elandsberg PNR (section 4.2.2) and the Stella Architect model interface (Land Management Decision-Support Tool). These end-products are intended to be useful for stakeholder groups who wish to engage in participatory scenario planning and disseminate further information on the SES problems. Furthermore, multi-stakeholder engagement to review model outcomes and further expert validation to refine the model accuracy could be addressed in future work (Process Step 13, Table 4.5) as a way to continue the social learning process and interactions with the land-use managers, to continue evolve the social network and move from analyses and recommendations to intervention (Clifford-Holmes et al., 2017b; Currie, 2018; Reed et al., 2010; Senge, 1990).

#### *4.2.3.2 Validation of the Ecological Model for biodiversity conservation*

An integral and highly important aspect of SD is model validation. Validation illustrated by dashed lines in Figure 4.4 show the iterative nature of validation throughout the five phases of the modelling process: during high-level systems mapping and qualitative modelling (Map); building the formal quantitative simulation model (Model); running scenarios and analyses (Simulate); and the design of policies or management guidelines (Design) (Barlas, 1996; Schwaninger and Groesser, 2009). Generic validation techniques employed in this study (Table 4.6) were considered in conjunction with recommendations by Groesser and Schwaninger (2012) to ensure model utility and effectiveness: (i) complexity hierarchy of validation tests; (ii) an integrative validation process; and (iii) a decision heuristic about when to stop formal validation efforts. Although an integrative validation process was not formally applied during this study, I intentionally adopted the attitude of continuously applying the validation tests throughout the modelling process. It is recognised that model validation is not entirely objective but is considered as "*usefulness with respect to some purpose*", and is therefore a non-technical, informal and qualitative process (Barlas, 1996). This is particularly true for types of omnipresent validation tests that were executed mentally without computational support such as structure and boundary assessment tests, parameter confirmation and the test of purpose (see tests marked with an asterisk (*) in Table 4.6).

Lastly, the decision on when to stop model validation processes, known as the validation cessation threshold (VCT), was also considered (

Figure 4.5 and Appendix 10.4) (Groesser and Schwaninger 2012). Several factors influenced the VCT – i.e. data intensity, modelling expertise of the modeller, the target group's SD experience and expectations of the model, model size and the risk associated with decisions (

Figure 4.5), and these factors are a result of their direct and indirect relationships (Appendix Figure 10.1). The best-case for an ideal modelling situation would be a very low VCT so that validation efforts cease relatively quickly in the modelling process. The interplay of these factors in the present inter- and transdisciplinary project and the resulting VCT of the model is articulated in section 7.2.4 of Chapter 7. Once the model was robust (i.e. all tests were employed and the simulation results sufficiently replicated the palaeo-reference modes), with a relatively low VCT, model validation ceased. The model was used to test the effects of different future grazing and burning regimes on vegetation composition (Process Step 11 and 12, Figure 4.4).

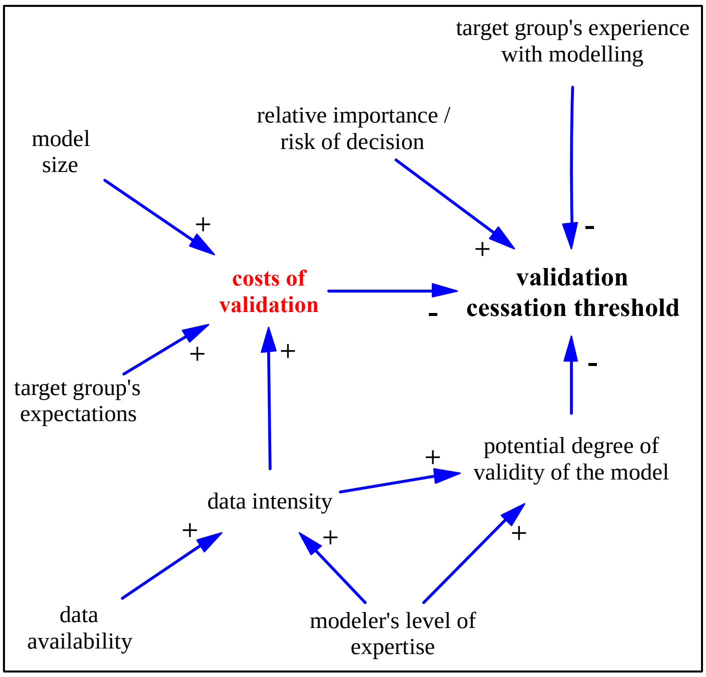



***Figure 4.5: Conceptual framework showing the determinants of the validation cessation threshold (VCT) (Groesser and Schwaninger, 2012).***

***Table 4.6: Summary of the various validation tests that were used to assess the Ecological Model for biodiversity conservation developed in this study (Sterman, 2000). Tests marked with an asterisk (*) can be executed without computational support.***

| Test | Purpose and requirements for passing validation test | Specific tools and procedures used in this study |
|---|---|---|
| **Structure and boundary assessment tests* | The model structure does not contradict the knowledge about the structure of the real-world system. | Model structure and boundary were compared with existing literature and reviewed with experts. |
| **Parameter confirmation test* | The parameter values reflect relevant descriptive and numerical knowledge of the system. All parameter values have real world equivalents. | Model parameter values were compared with literature and reviewed with experts. |
| *Dimensional consistency test* | All equations are dimensionally consistent without the use of parameters with units that have no real world meaning. | Model equations were inspected by carrying out unit analysis throughout model development using the modelling software (Stella Architect). |
| *Direct extreme condition test* | Each equation makes sense even when inputs take on extreme values. | Results were assessed for each flow equation when stocks were given imaginary max and min values. Responses to extreme values of each input (alone and in combination) were tested. |
| *Indirect extreme condition test* | The model responds plausibly when subjected to extreme policies, shocks, and parameters values. | Model behaviour was logically evaluated by running the model with extreme parameter values (large shocks and extreme conditions). |
| *Integration error test* | The results are not sensitive to the choice of time step or numerical integration method. | The time step was decreased, and a different integration method used (Eula to RK2 or RK4) to test for changes in model behaviour. |
| *Behaviour sensitivity analysis* | The purpose of the sensitivity analysis is to prioritise data collection effort and to identify leverage points for policy and scenarios analysis. | Model parameters were adjusted by +/- 25% and +/- 50% and the range of outputs generated were observed and recorded. |
| *Boundary adequacy test* | Important concepts for addressing the problem are endogenous to the model. | Multi-stakeholder engagement workshop and focused informal interviews were used to solicit expert opinion. A systems map, causal diagrams and model boundary charts were developed. The palaeo-record was analysed, and relevant literature reviewed. |
| *Historical behaviour* | When the model inputs are set at the historical values, the outputs will match the historical trend. | The simulation model results were compared with ca. 1300-year-old palaeoecological trends reported in Forbes et al. (2018). |
| **Test of purpose* | The model fulfils its purpose, e.g., whether it supports a decision about an investment policy (test of purpose) and whether the model purpose as given by the client/stakeholders has been achieved. | The broader sustainable land-use management problems were defined and the model purpose for Elandsberg PNR was clearly stated. |

# 5 CHAPTER 5: STATE CHANGE AT GROOT WINTERHOEK WILDERNESS AREA (GWWA) - APPLIED PALAEOECOLOGICAL RESULTS AND DISCUSSION

The historical events underlying the origin, maintenance and speciation mechanisms that lead to the remarkable plant diversity, and the time frame over which it occurred, have been the subject of considerable debate (Levyns, 1964; Schnitzler et al., 2011; Verboom et al., 2015, 2009). Topography, climate change and fire and grazing disturbance at long timescales have been overlain in more recent millennia by human management of landscapes. The San hunter-gatherers were in the region from at least 25 000 BP to the historical period, their hunting activities would have lightly influenced on marine and terrestrial ecosystem services in the region (Morris, 2018; Steele and Klein, 2005). The Khoikhoi pastoralists introduced livestock to the Western Cape approximately 2000 BP introducing an even greater land-use impact with the manipulation of vegetation using fire to stimulate pasture to support their livestock (Deacon, 1992; Humphreys et al., 1998). Thereafter, the arrival and colonisation by European settlers since the mid-17$^{th}$ century resulted in increased grazing pressure as well as the introduction of crop cultivation. European settlers also used a combination of fire suppression in some areas to protect agriculture and property, and regular burning in others. In the 20$^{th}$ century, agricultural intensification increased the impact on the environment in some areas, while in others nature reserves and stewardship schemes (section 3.1.2 and 3.4.1) aimed at conserving the CFR's unique biodiversity (Bergh and Visagie, 1985; Hoffman, 1997; Newton, 2008).

Furthermore, over the Holocene the CFR has experienced much climate variability: ranging from the warmer and dryer conditions during the Mid-Holocene altithermal (9000-5000 BP) (Meadows and Baxter, 1999) and the Medieval Climate Anomaly (MCA) (AD 900–1400), followed by the cold and wet Little Ice (LIA) (AD 1400–1800) (Nicholson et al., 2013) and the latter part of the 20$^{th}$ century characterised by a warming trend due to anthropogenic climate change (Cronin et al., 2003; Haensler et al., 2010). Future climate change poses a great threat with climate projections for the winter rainfall zone of the CFR suggesting an increase in aridity associated with a poleward drift of the prevailing westerly winds (Toggweiler and Russell, 2008), and a decrease of 10-30% in annual runoff by 2050 threatening both agricultural production and biodiversity in remaining areas of natural vegetation (Meadows, 2006; Turpie et al., 2002). Dramatic contractions of the Fynbos biome have been predicted in response to changing climate (Midgley and Thuiller, 2007; Midgley et al., 2005) but there is some uncertainty in these predictions (Hoffman et al., 2011) and more recent research shows a more optimistic outlook on vegetation response to climate change (Haensler et al., 2011) (section 3.1.2).

GWWA is an upland site that would have experienced less intense land-use than Elandsberg PNR and Rhenostervlei Farm. It is important in the context of the current study in providing a long-term record of response of Fynbos to climatic variability and in providing insights into the effects of land-use by San hunter-gatherers and KhoiKhoi pastoralists before European colonial settlement. Is there evidence from the GWWA006 palaeoecological record that biodiversity has responded to known past climatic variability? If so, what were the effects on ecosystem services such as plant biodiversity and soil erosion regulation? Are these ecosystem services and drivers such as climate, fire and herbivory are still within their historical range of variability? If critical ecological thresholds have already been crossed, is there evidence that ecosystem services are resilient to past change? Can the palaeo-record provide insights that may be useful to land-use managers in terms of fire and herbivory regimes necessary for biodiversity conservation and SES resilience at this important upland conservation site?

## 5.1 Historical Timeline of events: changes in direct and indirect variables

The GWWA has undergone extensive land-use change since the ca. AD 1800s which increased over time. A timeline that summarises long-term climatic and vegetation changes inferred from the literature, as well as direct land-use variables (i.e. relevant events that can be measured directly using palaeoecological data) and indirect land-use variables (i.e. other relevant events that cannot be inferred but are important in explaining this study's palaeoecological data) are shown in Table 5.1 as a basis for the interpretation of the GWWA006 data.

***Table 5.1: Timeline of events describing long-term climatic, vegetation and land-use change during last ca. 10 000 years. Direct variables include relevant events that can be measured using GWWA006 palaeoecological data, whereas indirect variables on land-use, climate and dominant vegetation are inferred from the literature, cannot be measured by the GWWA006 palaeo-data but are important in explaining the findings. Columns on the far left summarise the land-use types/groups, climate and dominant vegetation (green shading) during the respective time periods. As aisual snapshot, the climate column indicates red shading for warmer and drier climates and blue shading for wetter and colder climates. References are provided next to the text for each historical account with text specific to GWWA highlighted in bold (source: CapeNature, 2016). References in the climate column as specific to the climate.***

<table>
<tr><th>Land-use types/groups</th><th>Climate</th><th>Dominant vegetation</th><th>Variables with direct land-use impacts</th><th>Variables with indirect land-use impacts</th></tr>
<tr><td>Conservation<br>(1985 - present)</td><td rowspan="2">20th & 21st century warming (Haensler et al., 2011; Hoffman et al., 2011; Meadows, 2006; Turpie et al., 2002)</td><td>Restioid, ericoid, proteoid, asteraceous Fynbos with Cape thicket</td><td>• Dasklip Pass Road was tarred up to the Grootfontein turn off (AD 2003).<br>• Notable fires had a detrimental impact on the veld (AD 1993-1994).<br>• Maize (Zea mays) and grape vines (Viti vinivera) common in region (AD 1985) (Deacon 1986).</td><td>• GWWA received World Heritage Site status (2004)<br>• GWWA was proclaimed on 20 September 1985</td></tr>
<tr><td>Agricultural intensification<br>(1900-1985)</td><td>Increase in Asteraceous shrubs, decline in grasses</td><td>• Landowners perceive recent increased GWWA vegetation cover is linked to increased fire size and intensity since ca. 1970s (remote sensing data), and potential subsequent impacts on water supply and soil erosion (Holden et al. 2021).<br>• Notable storms left GWWA roads impassable, and areas flooded (AD 1962).<br>• Indigenous fynbos vegetation harvested (Protea spp) and sold to supplement income.<br>• Road constructed from Weltevrede to Mr. Roelof Wigboldus' farm at De Tronk (AD 1951)<br>-Building materials and equipment transported with donkeys and carts from Saron.<br>-Several crops cultivated: citrus, deciduous and nut trees (hazel, almond, pecans and walnuts), hectares of "bush tea", "buchu' (Agathosma betulina) and pasture crops (clovers, Serradella).<br>-Bat guano collected from caves around Die Hel used as fertilizer for cultivated lands.<br>• Elandsberg PNR showed increased E. rhinocerotis dominance, local fires and herbivory but fewer regional fires. Trends since ca. AD 1950s sustained through transition from agriculture to conservation in AD 1970s to present (Forbes et al. 2018).<br>• Frequent burning to support large-scale buchu harvesting and export trade (AD 1950).<br>• Notable storms left GWWA roads impassable and various areas flooded (AD 1941).<br>• Stock camp at Agterdam (AD 1936-1941).<br>• Settlement built, ca. half a hectare of Oak (Quercus) trees planted and land cultivated at Weltevrede (on Groot-Kliphuis River) by Engelbrecht family (AD 1936).<br>• Notable storms left GWWA roads impassable and various areas flooded (AD 1925)<br>• The van Huffel of Vischgat (Visgat) bought a portion of Perdevlei (AD 1913): Small areas of crops cultivated, fruit trees planted, and livestock farmed (sheep, goats and some cattle).</td><td>• Conservation of Agricultural Resources Act No. 43 of 1983 (Vester et al. 1992)<br>• Only five active farmers on the mountain (AD 1978-1984)<br>• Soil Conservation Act (Act No. 76 of 1969) Vester et al. 1992<br>• Department of Agriculture, Forestry & Water Affairs bought properties in the upper catchment of Vier-en-Twintig-riviere to conserve water resources (AD 1961-1978)<br>• The Dasklip Pass Road widened (AD 1960s)<br>• Porterville plateau road was extended from Zuurvlakte to Groot-Kliphuis and Perdevlei (AD 1951-1952)<br>• Soil Conservation Act No. 54 of 1964 (Vester et al. 1992).<br>• Settlements built at Groot Kliphuis, Langvlei and Perdevlei by the Engelbrecht family (AD 1941)<br>• The Dasklip Pass Road was built from Porterville, up the mountain onto the plateau to Weltevrede, Driebochfontein (AD 1936-1941)<br>• Wheat Importation Restrictions Act (Act No. 10 of 1930); Soil Erosion Adversary Council established (AD 1930) (Meadows 2003; Hoffman and Ashwell 2001)<br>• The Wigboldus brothers purchased the farm, De Tronk, and renamed it Groot Winterhoek (AD 1929-1930)</td></tr>
</table>

*(GWWA Historical Timeline continued – page 2/3)*

| Land-use types/groups | Climate | Dominant vegetation | Variables with direct land-use impacts | Variables with indirect land-use impacts |
|---|---|---|---|---|
| Agricultural intensification (1900-1985) | 20th Century warming | Increase in Asteraceous shrubs, decline in grasses | • **Mr. Retief bought De Tronk and permitted a small group of nomadic black Portuguese livestock farmers (Makatese) to build settlements (AD 1909).**<br>**-The Makatese cultivated a small area of crops (beans and tobacco), peach and Wattle (*Acacia* spp) trees; and kept a few sheep and goats.**<br>• Large indigenous herbivore, Quaqqa (*Equus quagga quagga*), became extinct (AD 1900s) (Krug et al. 2004; Krug and Krug 2007; Newton 2008). | |
| European settler agriculture (AD 1650-1900) | | | • Gum tree (*Eucalyptus*) seeds sold (AD 1853) (Wells et al. 1986).<br>• Introduction of *Acacia* spp by European colonialists (AD 1827-1835) (Neumann et al. 2011),<br>• **European Settlers used annual burning of veld to improve pasture for livestock (AD 1810).** | • **Permanent titles of "loan farms" properties registered after the second British annexation of the Cape (AD 1810).** |
| | Cold and wet Little Ice Age (LIA) (ca. AD 1400-1800)<br><br>LIA maximum cooling (AD 1700) Tyson et al. 2000<br><br>Wetter conditions from ca.1500 (Pakhuis Pass Shelter study) (Scott and Woodborne 2007 a,b) | | • Exponential increase in Asteraceous taxa, decline in Poaceae continuing to present (Klaarfontein Springs) 170 BP (*AD 1780) (Meadows and Baxter 2001). | |
| | | Increase in Asteraceous shrubs (incl. Renosterbos) and grasses. Exotics introduced (maize, pines, oats, wine grapes) | • Large indigenous herbivore, Blue buck (*Hippotragus leucophaeus*) hunted to extinction (AD 1700s) (Krug et al. 2004; Krug and Krug 2007; Newton 2008).<br>• Expansion of European colonial agriculture (AD 1700s) (Meadows and Sugden 1991).<br>• Increase in Asteraceae, grass and Renosterbos indicate drier conditions (since ca. 275 yr BP (*1675 AD)) (Cowling and Holmes 1992).<br>• Introduction of oaks (*Quercus*) and pines (*Pinus*) (±340 yr ago) (Neumann et al. 2011).<br>• Dutch planted maize (*Zea mays*) (AD 1656) (Wells et al. 1986).<br>• Dutch planted grape vine (*Viti vinivera*) (AD 1652-1662) (Bulpin and Rennie 1983).<br>• European colonialists cultivated indigenous edible plants, "waterblommetjies" (*Aponogeton* spp) (17th century) (Neumann et al. 2011).<br>• **Large indigenous herbivores became scarcer, hunters turned to stock farming. Hunting concession areas were converted to pasturing concession (AD 1652-1700s).**<br>• Increased charcoal possibly indicates uncontrolled burning (17th C) (Neumann et al. 2011). | • **Since Van Riebeek's Cape V.O.C. post, hunting reached Tulbagh and Vier-en-twintig riviere (AD 1652-1700s).** |
| Khoi-San hunter/herder (2000 BP-1600s) | | | • Shift from natural fynbos vegetation to grasslands - prominent decrease in Restionaceae and increase in Poaceae pollen (17th C) (Neumann et al. 2011).<br>• Wetting associated with the 'Little Ice Age' Holocene temperature minima resulted in a resurgence in fynbos abundance, but frequent fire driven by pastoralists appears to have reduced the fynbos community's functional diversity at the fynbos–succulent karoo biome boundary (Groenkloof, Namaqualand) (695-100 cal. yr BP) (MacPherson et al. 2018). | • Precipitation maxima in ca.600 cal BP (ca. AD 1350), ca.530 cal BP (ca. AD 1420) and ca.470 cal BP (ca. AD 1480) (Stager et al. 2012).<br>• Precipitation maxima (Stager et al. 2012) |
| | Warm and arid Medieval Climate Anomaly (MCA) (ca. AD 900-1400) Lee-Thorp et al., 2001; Nicholson et al., 2013; Stager et al., 2012) | Asteraceous shrubs, low grass abundance | • Asteraceous shrubs abundant and more regional fires, low levels of *E. rhinocerotis*, grasses, local fires and herbivory from ca. AD 750-1955. No major vegetation changes during MCA at Elandsberg PNR (Forbes et al. 2018).<br>• Decreased Poaceae pollen and increased micro-charcoal from 1900 BP attributed to Khoikhoi pastoralists and/or climate change since 1900 BP (Meadows and Baxter 2001). | |

*(GWWA Historical Timeline continued – page 3/3)*

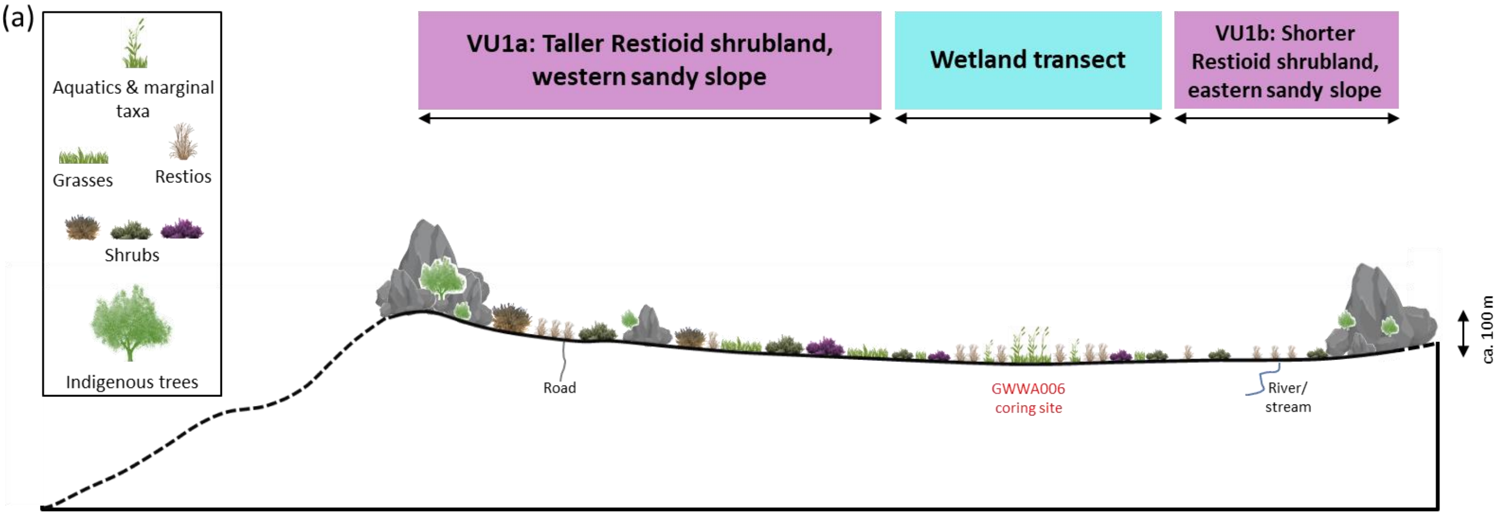
| Land-use types/groups | Climate | Dominant vegetation | Variables with direct land-use impacts | Variables with indirect land-use impacts |
|---|---|---|---|---|
| Khoi-San hunter/herder (2000 BP-1600s) | High precipitation (ca. AD 550-750) (Stager et al. 2012); Environment damp/wet (ca.1900-1000 cal. yr BP) and humid (ca.2000 cal. yr BP) (Neumann et al. 2011) | Fynbos shrubs (including Restios) and Afromontane Forest | • Increased disturbance as increased moisture was beneficial to herders (1800-2000 cal. yr BP) (Neumann et al 2011). Khoi-Khoi pastoralists were on the Vredenburg Peninsula by 1860 BP (Smith 1987; Meadows and Baxter 2001).<br>• Lowland Sand Plain Fynbos and restioid strandveld established amongst the low hills north of Klaarfontein, sheltered ravines around the sandstone hilltops were occupied by scrub and Afromontane Forest elements (*Olea*, *Podocarpus* and Santalaceae in pollen record) by 1900 BP) (Meadows and Baxter 2001). | |
| San hunter-gatherers (10 000-2000 BP) | Shift back to drier environment (2600–1900 cal. yr BP) (Neumann et al. 2011) | Grassy and Asteraceous Fynbos | • Dominance of an ambiguous grassy and Asteraceous fynbos. Summer drought and overall fynbos biome contraction, though woody fynbos shrubs persisted at Groenkloof, Namaqualand (4025-2005 cal. yr BP) (MacPherson et al. 2018).<br>• Asteraceous shrubland elements at Elands Bay Cave ca. 4300-3000 years ago (Parkington et al. 2000). | |
| | Wet (2800-2000 yr BP), humid and moist (4000-2000 yr BP) (Scott and Woodborne 2007; Meadows and Baxter 2000; Neumann et al.2011) | Fynbos | • Fynbos and fire thrived through summer moisture subsidies associated with sub-tropical easterly flow at Groenkloof, Namaqualand (5480-4025 cal. yr BP) (MacPherson et al. 2018). | |
| | Dry and hot Mid Holocene altithermal (MHA) (7700-4400 yr BP) (Neumann et al. 2011) | | | |
| | Increased humidity (7000 yr BP) (Martin 1968) | Fynbos, thicket and Afromontane Forest | • Fynbos-thicket mosaic (including *Dodonaea* and *Olea*) with Aizoaceae-type succulents and Asteraceae (Scott and Woodborne 2007a\|b). Increased *Podocarpus* and dry local vegetation ca. 7000 yr BP (Martin 1968).<br>• Mesic (moist) thicket vegetation 10 000-8000 years ago (Parkington et al. 2000). | |
| **GWWA006 core basal date at 144 cm - 10 647 to 10 706 cal BP (8698 to 8757 cal BC) at Pleistocene-Holocene boundary** | | | | |
| | Younger Dryas (YD) "Big freeze" cold period (13 000-11 500 yr BP) | Low shrubs, pure fynbos (increased Proteaceae) and Afromontane forest | • Dominant Proteoid fynbos elements at Elands Bay Cave 13 600-12 400 years ago) (Parkington et al. 2000). | • Pollen suggests stable climate ca. 14 000 cal yr BP (Meadows and Sugden 1991, 1993) |
| | Cold and moist Last Glacial Maximum (LGM) (26500-19000 yr ago) (Chase et al 2011) | | • Low shrubs (*Stoebe/Elytropappus*-type pollen) and "pure fynbos" (Ericaceae, *Passerina*, *Cliffortia* and Proteaceae taxa) at Pakhuis Pass Shelter during LGM (Scott and Woodborne (2007a,b)).<br>• Dominant Afromontane forest taxa at Elands Bay Cave during LGM (Parkington et al. 2000). | • More xeric with drier conditions at Boomplaas Cave (Deacon et al 1978). |

## 5.2 Vegetation survey and modern pollen estimation of GWWA

GWWA006 wetland basin is larger (diameter is ca. ±120-500 m) in comparison to the lowland agricultural site (Table 3.2). Thus, it was assumed that most of the fossil pollen would have originated extra-locally between 20-300 m away or regionally ca. 800 m away (Jacobson Jr and Bradshaw, 1981; Sugita, 1994) and suitable for detecting wetland and extra-local vegetation dynamics. However, since the GWWA006 coring site is an area of low-lying ground south of Zuurvlakte Farm with a sizable dam (Figure 3.2), it is subject to high velocity water flow and periodic scouring by flood water. Based on an adapted Braun-Blanquet vegetation survey protocol (van der Maarel, 2007), descriptions of the three vegetation units (VU1a, VU1b and wetland transect) in the vicinity of where the sediment core was retrieved are summarised in Table 5.2, with a detailed list in Appendix Table 10.3. Many of the 50 species that were identified during the modern vegetation survey are from families that are indistinguishable in the pollen record below family level (e.g. Proteaceae, Restionaceae, Poaceae, Cyperaceae, Asteraceae). 21 families and one genus (Asteraceae *Stoebe/Elytropappus*-type) could be identified in the pollen surface sample. Figure 5.1 uses artistic license to diagrammatically represent the vegetation profile of the study site with photographs shown in Appendix Figure 10.5 and Figure 10.6. There was a network of rivers/streams and erosion gullies indicating a general pattern of runoff flowing towards the larger wetland, thus showing evidence of seasonal waterlogging at the GWWA006 coring site (Figure 5.1).

***Table 5.2: Vegetation units at GWWA006 wetland, Groot Winterhoek Wilderness Area study site.***

| Vegetation Unit | Description | Dominant taxa |
|---|---|---|
| VU1a: Taller Restioid shrubland, western sandy slope | Western upland margin of wetland, outcrops of sandstone and patchy restioid fynbos with taller shrubland (ca. 50 cm high), on gentle sandy slope comprising ca. 14% bare ground. | *Metalasia* spp (Asteraceae), Poaceae, Restionaceae and *Cliffortia ruschifolia* (Rosaceae). Rocky outcrops act as fire refugia for Afromontane Forest-type, *Heeria argentea* (Anacardiaceae), and Cape Thicket taxa *Searsia* spp (Anacardiaceae) and *Maytenus oleoides* (Celastraceae). |
| VU1b: Shorter Restioid shrubland, eastern sandy slope | Eastern upland margin of wetland, outcrops of sandstone and patchy restioid fynbos with short shrubs on sandy soils comprising ca. 45% bare ground. | Several Restio *spp* and *Stoebe* spp (Asteraceae). Rocky outcrops act as fire refugia for tall shrubs or small trees: *Podalyria* spp (Fabaceae), *Myrsine africana* (Primulaceae), *Diospyros* spp (Ebenaceae), *Searsia* spp (Anacardiaceae), *Dodonaea viscosa* (Sapindaceae), *Maytenus oleoides* (Celastraceae). |
| GWWA006 Wetland transect | The transect represents a gradient in plant communities associated with wetland characteristics consisting of six dominant sections from the wetland centre where GWWA006 core was retrieved to the periphery in a westward direction. Note that water flow decreased from section one to six: | 1. *Typha* (Typhaceae); 2. *Anthochortus crinalis* (Restionaceae); 3. Cyperaceae and Asteraceae; 4. *Restio* spp and *Marchantia polymorpha;*5. *Restio* spp, and 6. *Stoebe* spp, *Erica* spp, *Cliffortia (other)*, *Drosera* with several different grasses also present. There is an old packing shed north-west of the wetland and evidence of disturbance (dams and rubble) where *Cliffortia ruschifolia* were present. |



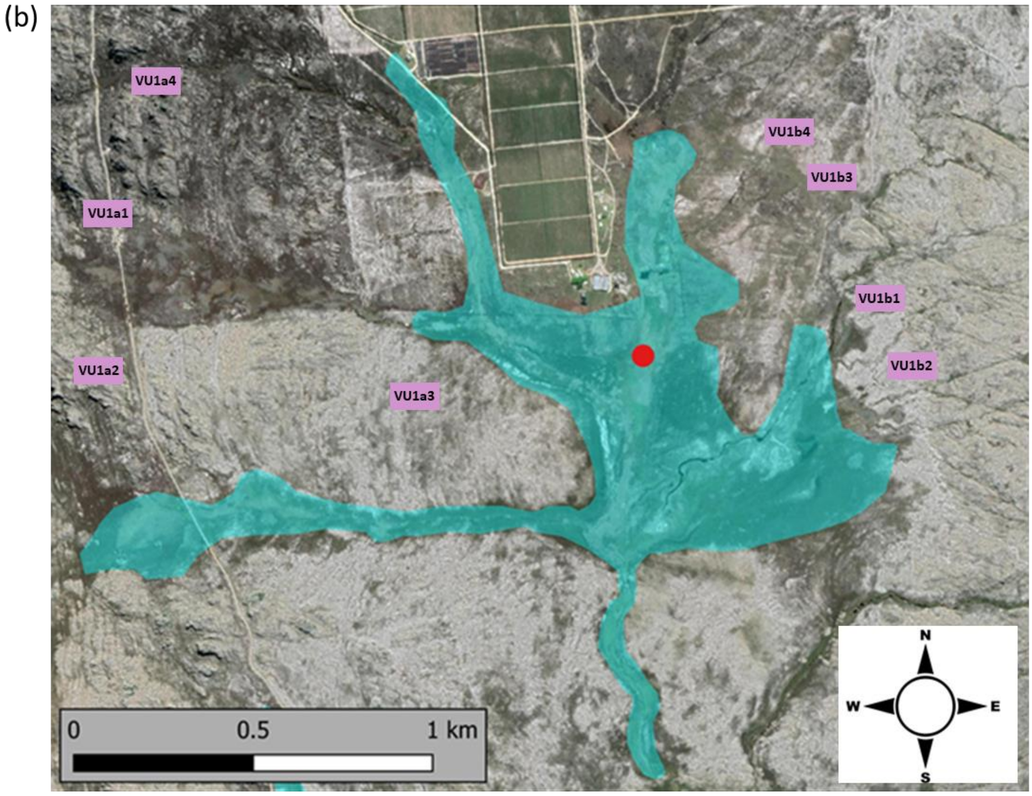




*Figure 5.1. (a) Diagrammatic representation of the profile of vegetation units (VU 1a and 1b) that were surveyed. Google earth map of selected area of GWWA in 2018. Demarcated areas in blue represents the wetland and pink annotations indicate vegetation units that were chosen according to the most dominant vegetation type during the vegetation survey conducted in 2020. GWWA006 is ca. 0.22 km away from a dam at Zuurvlakte Farm in the north at 946 m above sea level.*

## 5.3 Chronology of GWWA006 - Bacon Age-Depth model

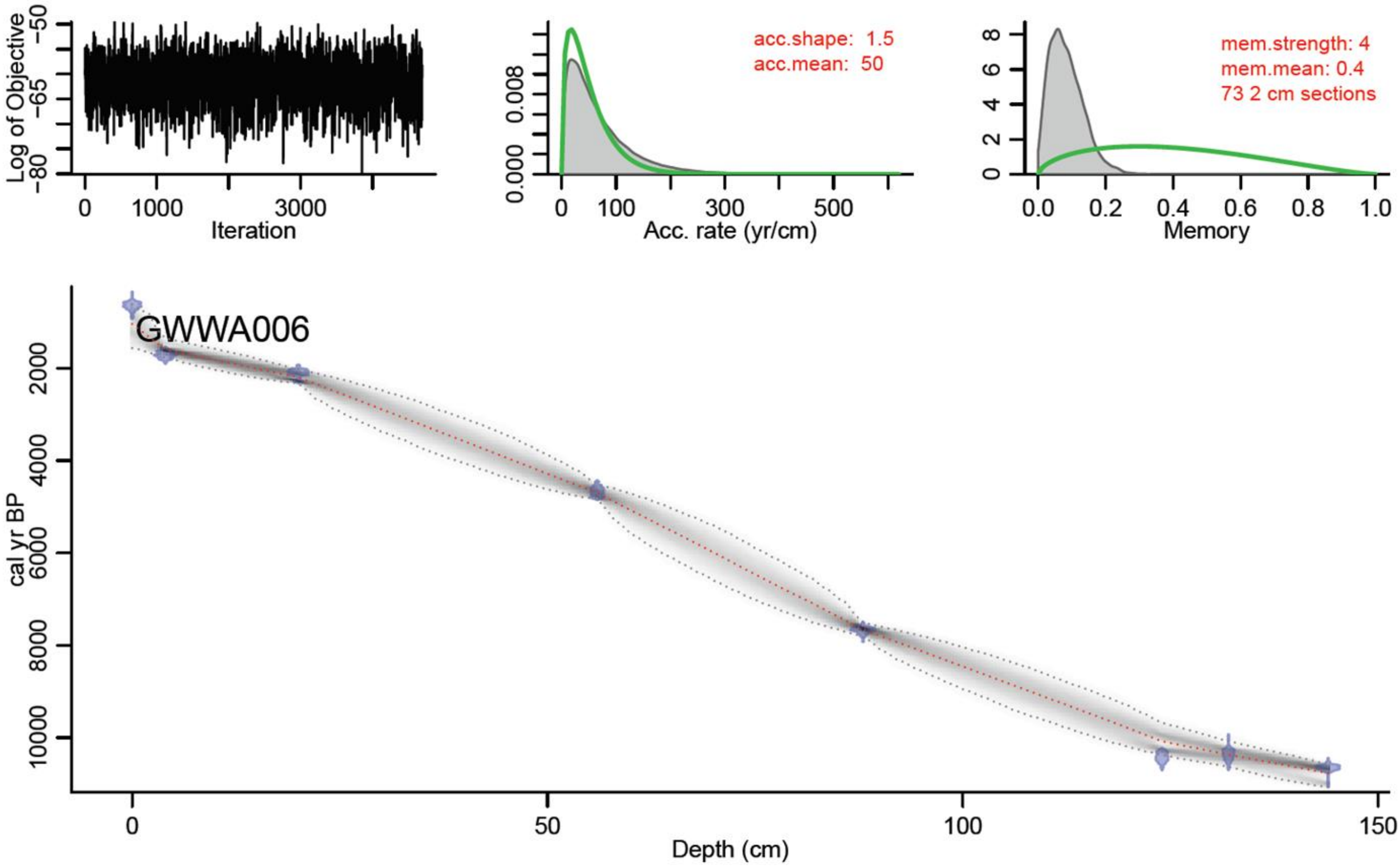



***Figure 5.2: Age-depth model for samples from 0 to 144 cm for the GWWA006 core including eight samples that were dated using AMS radiocarbon dating and calibrated using SHCal20 (Hogg et al., 2020).***

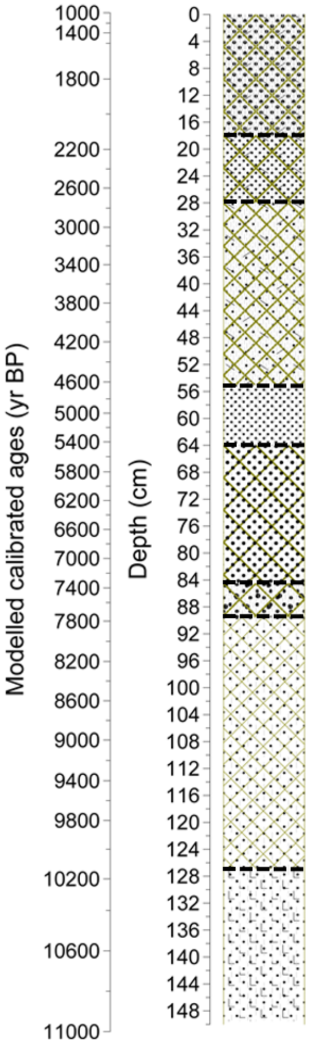
To determine the chronology of environmental reconstruction, an age-depth model was produced with the eight AMS radiocarbon dates using Bacon in R (Figure 5.2) (Blaauw and Christen, 2013). The raw dates, errors and calibrations are shown in Appendix Table 10.4. The top 144 cm of the core are shown. The age at the base of this section of the core was ca. 10 765 cal yr BP. Constraining the age-depth model through a modern surface date showed an unfeasibly fast accumulation rate at recent times, therefore the age-depth model was not constrained in this way and suggested a surface date of ca. 1030 cal yr BP. This type of age-depth model is common when the top of the core comprises unconsolidated sediment. The sedimentation appeared to show no reversals and the accumulation rates are reasonable, once the surface age was unconstrained. Average sediment accumulation rates showed a decreasing trend over time: a consistent and steady sediment accumulation (0.015 cm/year) from 144-53 cm; a slightly slower sedimentation rate yet still consistent (0.014 cm/year) from 53-30 cm; followed by a higher sediment accumulation rate (0.021 cm/year) from 30-4.5 cm; decreasing to a significantly lower sediment accumulation rate (0.007 cm/year) in the upper depths from 4.5-0 cm.

## 5.4 Sediment description

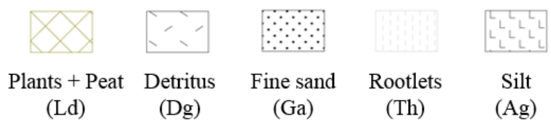
The GWWA006 sediment core was retrieved from a 240 m diameter basin within a larger wetland complex of ±120-500 m diameter (section 3.3). The total drilling length was 401 cm, with 105 cm compression. Thus, the total length of the sediment core retrieved was 296 cm. During transport and storage of the sediment core, water and sediment was lost from the core between 150 cm and 296 cm and therefore it was assumed that that section of sediment was compromised for palaeoecological analysis. Given the AMS radiocarbon dating at 144 cm resulting in a basal age of ca. 10 765 BP and the five sub-sampled sections analysed for range finding, only the top 144 cm was analysed for palaeoenvironmental proxies. The impacts of compression on the GWWA006 palaeo-record were taken into account. Perhaps there would have been higher temporal resolution if there was less compression but as evidenced by the age-depth model showing no reversals or mixing, the chronology is intact for reliable analysis of relative changes in palaeo-proxies.

Results for sediment composition (Troels-Smith, 1955) and Loss on Ignition ($LoI_{550}$) (Bengtsson and Enell, 1986; Heiri et al., 2001; Walter E. Dean, 1974) can be found in Table 5.3 and Figure 5.7, respectively. The sediment composition of waterlogged samples of the GWWA006 core consisted of two different shades of black (Munsell, 1942), described according to eight sections consisting mostly of fine sand, macro-plant material and peat. Section 55-64 cm was distinctly different and consisted mostly of fine sand with rootlets, which may relate to changes in geochemistry and physical properties in the core. Sediments below 144 cm were of similar composition to that of the top 144 cm.

***Table 5.3: Troels-Smith stratigraphy and Munsell soil colour description of sediments in the GWWA006 core. White dashed line shows the gradational upper contact.***




| Section depth (cm) | Troels-Smith Description | Munsell soil colour and code | Notes |
|---|---|---|---|
| 0-18 | Ld 1; Ga 2; Dg 1; Th +4 | Black (2.5Y, 2.5/1) | Peaty, no clay with herbaceous and ligneous pieces. |
| 18-28 | Ga 3; Dg 1; Ld +1; Th+3 | Black (5Y, 2.5/1) | Very little peat, no clay. |
| 28-55 | Ld 2; Ga 1; Dg 1; Th+4 | Black (5Y, 2.5/1) | Very peaty, no clay. |
| 55-64 | Ga 3; Th 1 | Black (2.5Y, 2.5/1) | Actually Ga 3.5, some herbaceous pieces but less than 25%. |
| 64-84.5 | Ga 2; Ld 1; Th 1 | Black (5Y, 2.5/1) | Herbaceous pieces but less than 25%. |
| 84.5-89.5 | Ga 1; Gs 2; Ld 1; Th +4 | Black (5Y, 2.5/1) | Coarser sand, peaty. |
| 89.5-127 | Ga 1; Ld 2; Th 1 | Black (5Y, 2.5/1) | Herbaceous pieces but less than 25%. |
| 127-149.5 | Ag 2; Th1; Ga 1 | Black (5Y, 2.5/1) | |



## 5.5 Palaeoenvironmental proxy results for GWWA006 core

### 5.5.1 Pollen results for GWWA006: A proxy for plant biodiversity

#### *5.5.1.1 GWWA006 Pollen Zonation*

Table 5.4 summarizes the pollen zonation results that are illustrated in Appendix Figure 10.8 and Figure 10.9. Zonation depicts the distribution of percentage and concentration pollen types based on depth of the GWWA006 core. The broken stick model (Appendix Figure 10.8) indicated that there were two significant zones because the value of the black line was above the critical value represented by the red line. Despite the lack of statistical significance in the upper depths (0-53 cm), three sub-zones are illustrated with a dashed line and used to describe changes in the pollen assemblage based on the square root transformed diatom data and the Bray-Curtis similarity data.

***Table 5.4: Details of zonation for the GWWA006 pollen record. The solid line separating Zones G1 and G2 indicates statistically significant zonation, while the dashed lines indicate sub-zones within G2.***

| Pollen zone | Number of sub-sample depths analysed | Depth ranges of samples (cm) | Dates of pollen zone boundaries (cal yr BP) | Number of years |
|---|---|---|---|---|
| Sub-zone G2c | 9 | 0-4.5 | ca. 1030-1610 | ca. 580 |
| Sub-zone G2b | 16 | 4.5-30 | ca. 1610-2885 | ca. 1275 |
| Sub-zone G2a | 10 | 30-53 | ca. 2885-4495 | ca. 1610 |
| Zone G1 | 18 | 53-144 | ca. 4495-10 765 | ca. 6270 |

On average, within the ca. 10 765-1030-year timescale of the GWWA006 pollen spectrum, the most abundant pollen was that of *Cliffortia* spp (mean 25% and mean 43 283 grains/cm$^3$) characteristic of the Fynbos vegetation grouping. The second most abundant pollen taxon in the GWWA006 record was Poaceae (mean 22% and mean 26 757 grains/cm$^3$), representative of the grassy element of the Fynbos landscape. Other pollen types characteristic of Fynbos vegetation were also relatively abundant: Ericaceae pollen (mean 13% and mean 16 073 grains/cm$^3$), Asteraceae long-spine type-1 pollen (mean 11% and mean 12 522 grains/cm$^3$), and Restionaceae pollen (mean 9% and mean 10 228 grains/cm$^3$). Pollen representative of Thicket/Forest vegetation were less abundant, yet variability in the trend over time yielded interesting results. *Maytenus oleoides* (Celastraceae) was the most abundant taxon (mean 0.7% and mean 804 grains/cm$^3$) characteristic of Thicket/Forest vegetation. Changes in the dominance of these vegetation types that occurred over time are described below (Figure 5.3 and Figure 5.4). Unknown pollen grains were relatively low (mean 10% and 12 095 grains/cm$^3$) in abundance as most pollen grains were identifiable in the GWWA record.

#### *5.5.1.2 Pollen Zone G1 (18 sub-sample depths; 144-53 cm; ca. 10 765 - 4495 cal yr BP)*

In Zone G1, *Cliffortia* (representative of Fynbos vegetation) was the most dominant (mean 45% and 100 895 grains/cm$^3$). *Cliffortia* pollen increased gradually, decreased until ca. 7660 (88 cm) and then increased until it reached its highest abundance of 64% in ca. 6927 BP (80 cm) and 269 267 grains/cm$^3$ in ca. 5806 BP (68 cm). Thereafter it declined in the later part of Zone G1. Poaceae pollen representative of the grassy element of the Fynbos landscape was the next most dominant (mean 17% and 30 631 grains/cm$^3$) and followed a similar fluctuating pattern to *Cliffortia* pollen, which was more prominent in the concentration data. Additional pollen types characteristic of Fynbos vegetation were also abundant: Ericaceae (mean 10% and 17 189 grains/cm$^3$), followed by Restionaceae (7% and 12 503 grains/cm$^3$), and Asteraceae (mean 5% and 9 373 grains/cm$^3$). The restioid-proteoid-ericoid abundance in ca. 9016-8193 BP (108-96 cm) shifted to restioid-ericoid and *Cliffortia*-abundant in ca. 7659-5447 BP (88-64 cm). Pollen characteristic of the Thicket/Forest vegetation sub-type was low in abundance (mean 0.6% and 1035 grains/cm$^3$) with *Podocarpus* (mean 0.2% and 483 grains/cm$^3$) and *Maytenus oleoides* (mean 0.3% and 457 grains/cm$^3$) present in low abundance.

Additionally, the ratio of aquatics and marginal:terrestrial pollen was relatively low with a mean ratio of 0.23. However, it reached its highest value (ratio of 0.7) in ca. 4700 BP (56 cm). Typha/Phragmites-type pollen was the most abundant (29 620 grains/cm$^3$) followed by Cyperaceae and Juncaceae (Figure 5.4). A significantly high relative abundance in degraded pollen (38 664 grains/cm$^3$) occurred in ca. 5447 BP (64 cm).

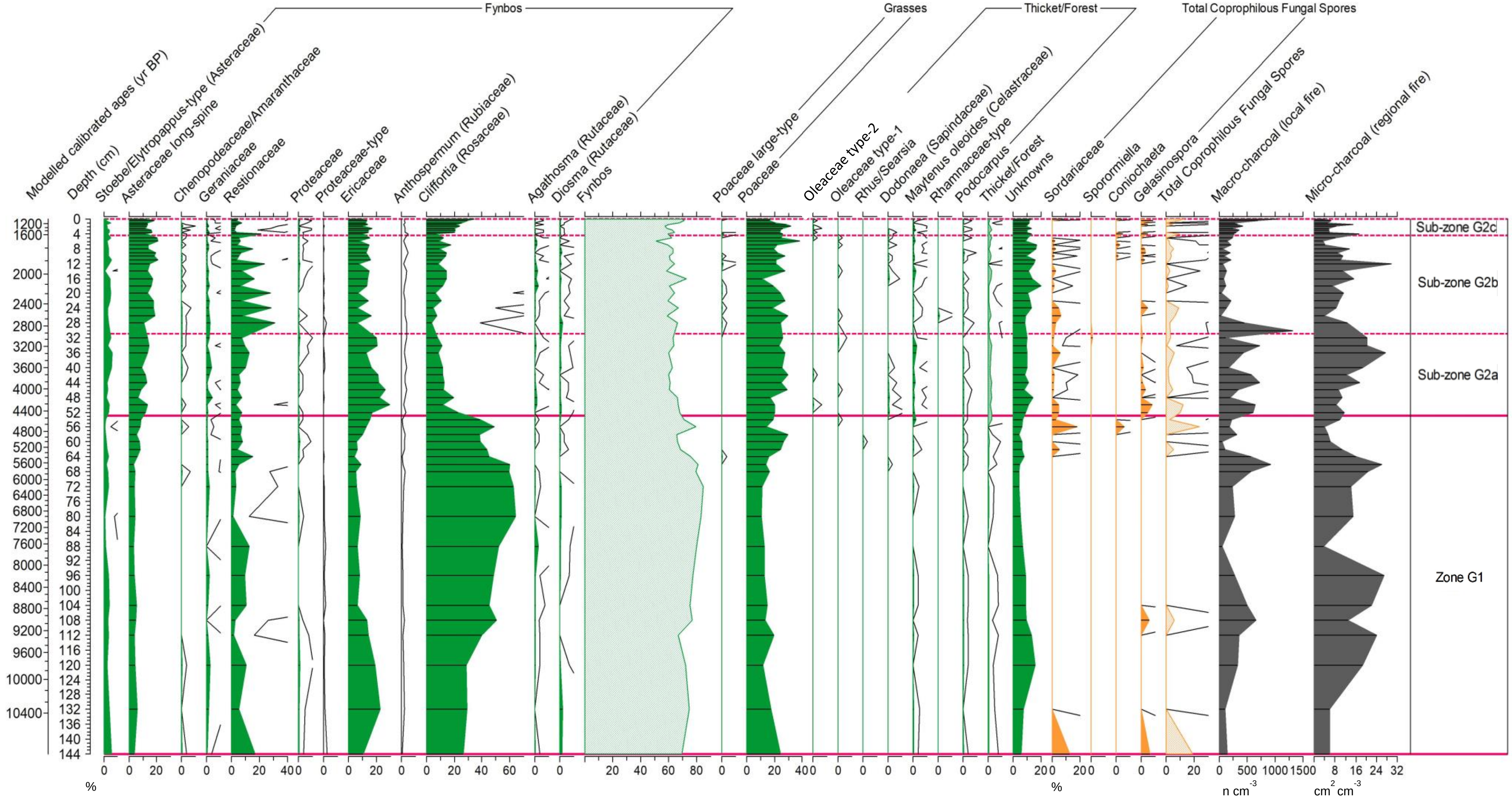
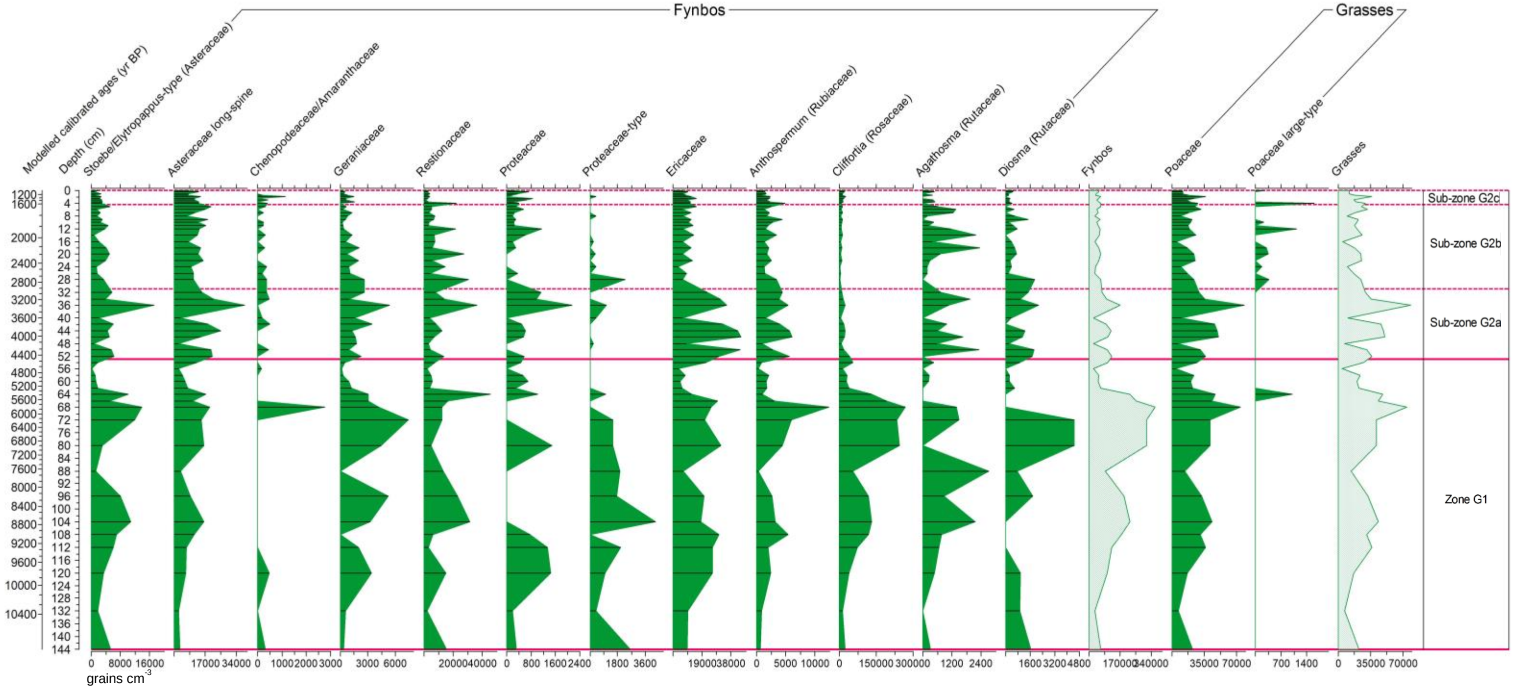



***Figure 5.3:*** ***Percentage pollen diagram with selected palaeo-proxies (coprophilous fungal spores, and macro- and micro-charcoal) from the GWWA006 core retrieved at Groot Winterhoek Wilderness Area, Middle Berg River Catchment, South Africa. The pollen sum included terrestrial and unknown pollen. x10 exaggeration is shown for low pollen abundances. Summary graphs with pollen types grouped into ecological groupings of major vegetation types. Zonation is calculated by cluster analysis based on 70% minimum resemblance levels – statistically significant zones are indicated by the solid line (Zone G1 and Zone G2) whereas dashed lines show sub-zones (G2a, G2b, G2c). Modelled calibrated ages using Bacon are shown on the left.***



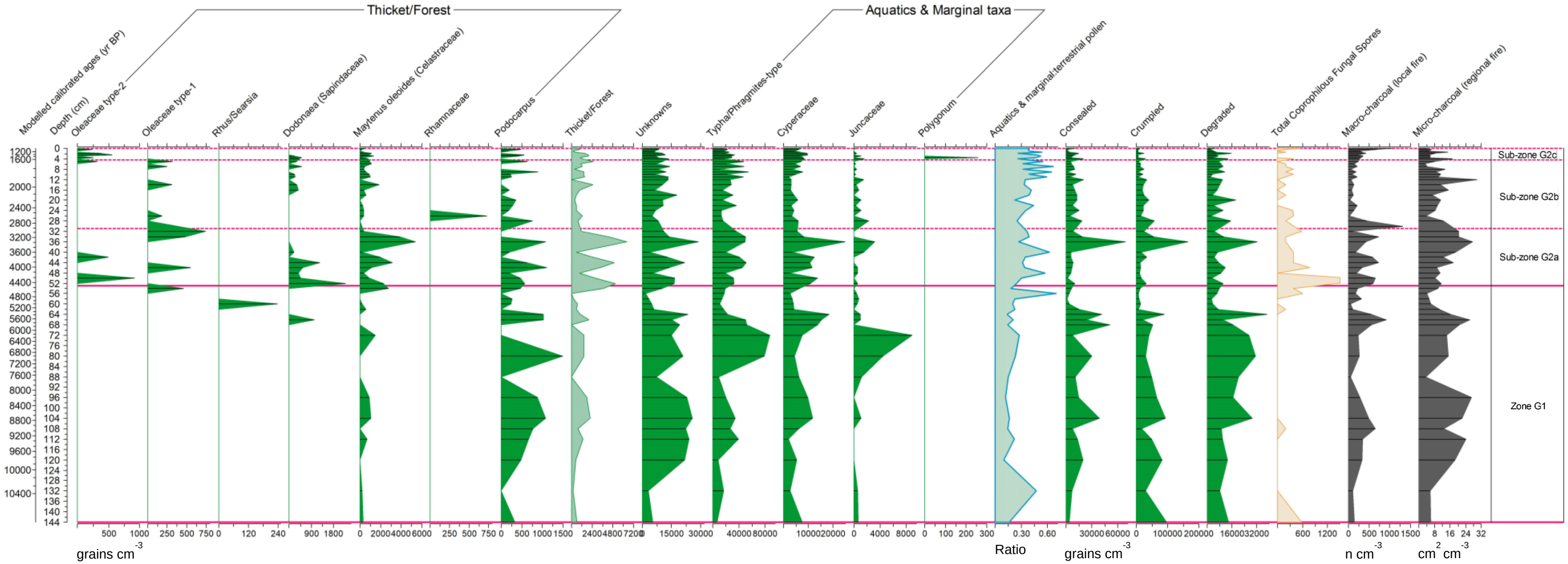




***Figure 5.4: Concentration pollen diagram with coprophilous fungal spores, and macro- and micro-charcoal from the GWWA006 core. Pollen types are grouped into major vegetation types. Statistically significant zones are indicated by the solid line whereas dashed lines show sub-zones (G2a, G2b, G2c). Modelled calibrated ages are shown on the left.***

#### *5.5.1.3 Pollen Sub-zone G2a (10 sub-sample depths; 53-30 cm; ca. 4495-2885 cal yr BP)*

In Sub-zone G2a, Poaceae pollen (representative of the grassy element of the Fynbos landscape) was the most dominant (mean 25% and 37 430 grains/cm$^3$). Poaceae pollen fluctuated, seen by several peaks and troughs throughout the sub-zone. This pattern was more prominent in the concentration data likely due to the Fagerlind effect in the percentage data. For the concentration data only, the greatest abundance of Poaceae pollen was 78 436 grains/cm$^3$ in ca. 3318 BP (36 cm). Ericaceae pollen representative of the Fynbos vegetation sub-type was the next most dominant (mean 21% and 29 870 grains/cm$^3$) and followed a similar fluctuating trend to grasses and Thicket/Forest pollen. Additional pollen types characteristic of Fynbos vegetation were also abundant: *Cliffortia* (mean 13% and 18 540 grains/cm$^3$), followed by Asteraceae (mean 12% and 18 140 grains/cm$^3$) and Restionaceae (7% and 11 197grains/cm$^3$). Although pollen typical of Thicket/Forest vegetation sub-type generally remained low in abundance (mean 2.1% and 3212 grains/cm$^3$), the trend increased during Sub-zone G2a with *Maytenus oleoides* being the most abundant as it reached its greatest abundance of 3% and 5370 grains/cm$^3$. The ratio of aquatics and marginal:terrestrial pollen increased to a mean ratio of 0.38 with some variability. Typha/Phragmites-type pollen remained the most abundant (37 920 grains/cm$^3$) (Figure 5.4). Degraded pollen was relatively lower in Sub-zone G2a, however, a high abundance (32 220 grains/cm$^3$) occurred in pollen occurred in ca. 3318 BP (36 cm).

#### *5.5.1.4 Pollen Sub-zone G2b (16 sub-sample depths; 30-4.5 cm; ca. 2885-1610 cal yr BP)*

In Sub-zone G2b, Poaceae pollen remained the most dominant pollen type over time (mean 24% and 19 620 grains/cm$^3$). It continued to fluctuate throughout the sub-zone and for the percentage data only, the greatest abundance of Poaceae pollen was 39% in ca. 1666 BP (6 cm), just before the Sub-zone G2b/G2c transition. Asteraceae long-spine pollen representative of the Fynbos vegetation sub-type was the next most dominant (mean 17% and 13 303 grains/cm$^3$) and the abundance continued to fluctuate. Similar to previous parts of the GWWA006 core, additional pollen types characteristic of Fynbos vegetation were abundant: Restionaceae (mean 13% and 9972 grains/cm$^3$), followed by Ericaceae (mean 12% and 9561 grains/cm$^3$) and *Cliffortia* (11% and 8483 grains/cm$^3$). Thicket/Forest pollen decreased slightly (mean 1.4% and 1075 grains/cm$^3$) but *Maytenus oleoides* remained the most abundant taxa of this sub-type.

#### *5.5.1.5 Pollen Sub-zone G2c (9 sub-sample depths; 4.5-0 cm; ca. 1610-1030 cal yr BP)*

In Sub-zone G2c, Poaceae pollen that is representative of the grassy element of the Fynbos landscape remained dominant (mean 24% and 19 839 grains/$cm^3$) with a similar fluctuation as the previous sub-zone. During this period, *Cliffortia* pollen characteristic of Fynbos vegetation sub-type was the next most dominant (mean 23% and 17 416 grains/$cm^3$). It showed a steep increasing trend over time which was more prominent in the percentage data. Like previous sub-zones, additional pollen types characteristic of Fynbos vegetation were abundant: Asteraceae long-spine pollen (mean 17% and 13 303 grains/$cm^3$) initially decreased and then increased in the later part of the Sub-zone G2c. Ericaceae increased slightly (mean 13% and 10 089 grains/$cm^3$) but followed a similar fluctuating trend as before. Restionaceae (mean 13% and 9972 grains/$cm^3$) was high in abundance initially and then decreased during Sub-zone G2c. Thicket/Forest pollen (mean 1.3% and 1082 grains/$cm^3$) remained stable at a low level with *Maytenus oleoides* as the most abundant taxa of this sub-type.

### 5.5.2 Macro-charcoal and micro-charcoal as proxies for local and regional fire

In Zone G1 (ca. 10 765-4495 BP; 144-53 cm), macro-charcoal was on average 325 particles/$cm^3$. It showed a slight increasing trend in early parts of the zone with a peak (653 particles/$cm^3$) in ca. 9015 BP (108 cm) before it declined (59 particles/$cm^3$) halfway through Zone G1 in ca. 7660 BP (88 cm). The trend increased again to a higher peak (920 particles/$cm^3$) in ca. 5630 BP (66 cm) before decreasing and increasing again in the later part of the zone. Micro-charcoal was on average 13.8 $cm^2/cm^3$ in G1. Despite an inverse trend occurring in ca. 9015 BP (108 cm) when micro-charcoal was low whilst macro-charcoal was high, micro-charcoal followed a similar fluctuating pattern during G1. Micro-charcoal experienced a coinciding trough (3.6 $cm^2/cm^3$) with macro-charcoal in ca. 7660 BP (88 cm) and coinciding peak (26.2 $cm^2/cm^3$) in ca. 5630 BP (66 cm).

In Sub-zone G2a (ca. 4495-2885 BP; 53-30 cm), macro-charcoal was on average the highest (mean 454 particles/$cm^3$) compared to the other sub-zones. It continued to show a fluctuating trend throughout G2a. Peaks occurred in ca. 4290 BP (50 cm) (646 particles/$cm^3$), ca. 3880 BP (44 cm) (726 particles/$cm^3$) and ca. 3173 BP (34 cm) (724 particles/$cm^3$). Macro-charcoal troughs occurred in ca. 4160 BP (48 cm) (202 particles/$cm^3$), ca. 3600 BP (40 cm) (173 particles/$cm^3$) and ca. 3025 BP (32 cm) (121 particles/$cm^3$). The relative average abundance of micro-charcoal was also the highest (mean

15.8 $cm^2/cm^3$) in Sub-zone G2a. It showed a steady increasing trend to a peak of 27.4 $cm^2/cm^3$ in ca. 3320 (36 cm) before decreasing in the later part of G2a.

In Sub-zone G2b (ca. 2885-1610 BP; 30-4.5 cm), macro-charcoal was on average the lowest (mean 220 particles/$cm^3$) compared to the other sub-zones. It experienced an unprecedented high (1311 particles/$cm^3$) in ca. 2890 BP (30 cm) at the G2a/G2b transition followed by an unprecedented low of 25 particles/$cm^3$ in ca. 2615 BP (26 cm) in the GWWA006 record. The abundance of macro-charcoal abundance remained low, increasing very slightly towards the end of Sub-zone G2b. The relative average abundance of micro-charcoal was moderate (mean 10.6 $cm^2/cm^3$) in Sub-zone G2b. Like macro-charcoal, it decreased initially but followed a different increasing trend to an unprecedented high of 29.8 $cm^2/cm^3$ in ca. 1890 BP (12 cm) before decreasing in the later part of G2a to an unprecedented, and its lowest, relative abundance of 3.4 $cm^2/cm^3$ in ca. 1665 BP (6 cm) in the time period in the sediment sequence from this site.

In Sub-zone G2c (ca. 1610-1030 BP; 4.5-0 cm), macro-charcoal was on average the second highest (mean 443 particles/$cm^3$). It increased steeply during the sub-zone to a high of 1059 particles/$cm^3$ in ca. 1030 BP (0 cm). With the lowest relative abundance of micro-charcoal occurring in Sub-zone G2c (8.3 $cm^2/cm^3$), micro-charcoal's trend was more variable. Noticeable peaks occurred in ca. 1590 BP (4 cm) (17.3 $cm^2/cm^3$) and ca. 1160 BP (1.5 cm) (14.9 $cm^2/cm^3$) and troughs in ca. 1290 BP (2.5 cm) (15.7 $cm^2/cm^3$) and ca. 1030 BP (0 cm) (4.2 $cm^2/cm^3$).

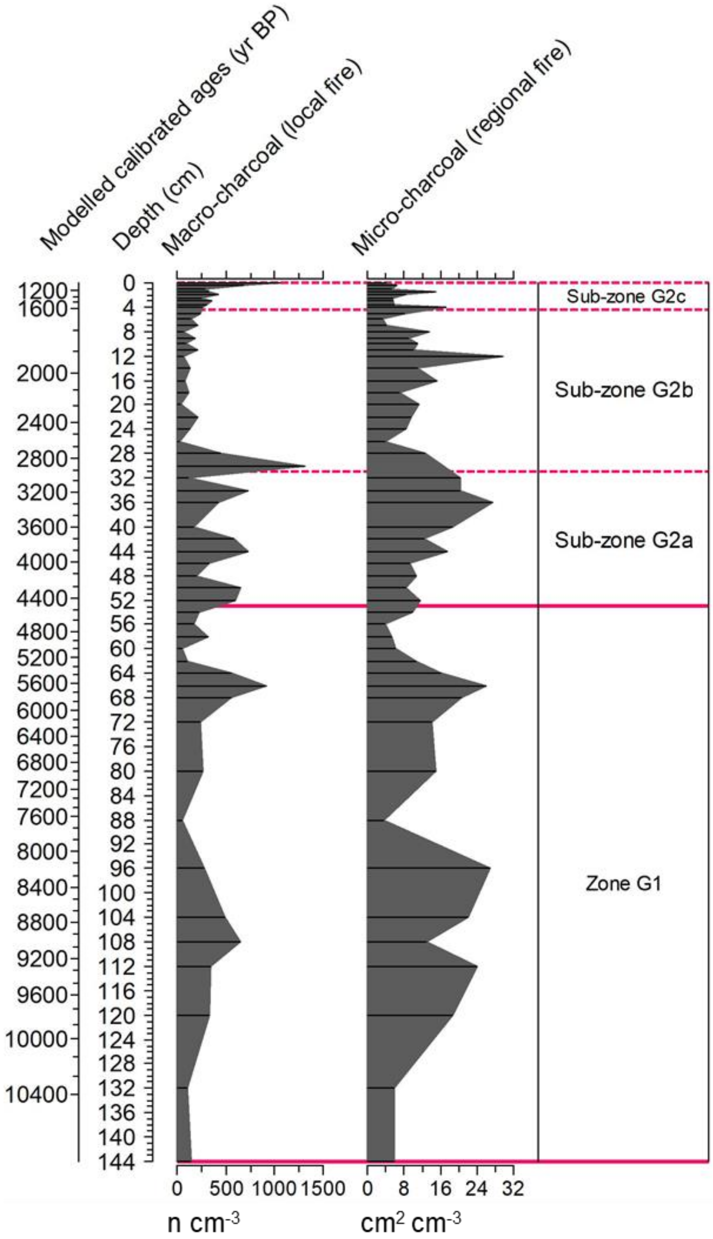



*Figure 5.5: Macro-charcoal (local fire) and micro-charcoal concentration (regional fire) for the GWWA006 core.*

### 5.5.3 Spores

#### *5.5.3.1 Coprophilous fungal spores as a proxy for herbivory*

In Zone G1 (ca. 10 765-4495 BP; 144-53 cm), the mean of aggregated coprophilous fungal spores (the sum of *Coniochaeta,* Sordariaceae, *Sporormiella* and *Gelasinospora*, as an indicator of relative herbivory abundance) was low (mean 3.1% and 109 grains/cm$^3$) since they were mostly absent, except for a low abundance of Sordariaceae and *Gelasinospora* in ca. 10 765 BP (144 cm), and the presence of *Gelasinospora* in ca. 9015 BP (108 cm). Sordariaceae and *Coniochaeta* occurred towards the later part of G1, with an unprecedented peak in Sordariaceae owing to the highest aggregated coprophilous fungal spores (24%) in 4700 BP (56 cm) in the time period in the sediment sequence from this site. This high peak was only evident in the percentage spore data, possibly due to the Fagerlind effect.

In Sub-zone G2a (ca. 4495-2885 BP; 53-30 cm), the mean of aggregated coprophilous fungal spores increased to its highest average (mean 4.2% and 599 grains/cm$^3$), with a fluctuating trend that generally decreased throughout the sub-zone. The higher relative abundance was more apparent in the concentration data compared to the percentage data. Thus, the highest abundance of the aggregated coprophilous fungal spores was in G2a for the concentration data (1547 grains/cm$^3$) in ca. 4290-4425 BP (50-52 cm), whereas the highest abundance was in G1 for percentage data. Concerning individual coprophilous fungal spore types, Sordariaceae remained the most dominant taxon (mean 2.1% and 271 grains/cm$^3$), followed by *Gelasinospora* (mean 1.9% and 309 grains/cm$^3$). *Coniochaeta* was absent during this sub-zone while *Sporormiella* occurred for the first and only time just before the G2a/G2b transition in ca. 3025 BP (32 cm).

In Sub-zone G2b (ca. 2885-1610 BP; 30-4.5 cm), the mean of aggregated coprophilous fungal spores was relatively lower (mean 2.6% and 180 grains/cm$^3$), with a trend of slight fluctuation at a low level throughout G2b. Sordariaceae remained the most dominant taxon (mean 1.5% and 108 grains/cm$^3$), and *Gelasinospora* (mean 0.8% and 48 grains/cm$^3$) the second most abundant. *Coniochaeta* was present at a low level towards the later part in ca. 1815 BP (10 cm) and ca. 1705 BP (7 cm).

In Sub-zone G2c (ca. 1610-1030 BP; 4.5-0 cm), on average the mean of aggregated coprophilous fungal spores as an indicator of relative herbivory abundance increased in the percentage spore data (mean 4%) but decreased in the concentration data (150 grains/cm$^3$). However, trends were similar as the level of coprophilous fungal spores remained low and increased slightly over time.

#### *5.5.3.2 Non-Coprophilous Spores*

In Zone G1 (ca. 10 765-4495 BP; 144-53 cm), the average of aggregated abundance of non-coprophilous spores (Trilete, Monolete and other non-coprophilous spores) showed a high mean in the percentage spore data (96.9%) but a low average concentration (3543 grains/cm$^3$) relative to the other zones. This discrepancy between the different datatypes is observed throughout the GWWA006 spore record i.e. even though concentration is low, the non-coprophilous spores were more abundant than the coprophilous spores and therefore constitute a high percentage. Monolete spores were less common (mean 2% and 32 grains/cm$^3$) than Trilete (mean 15% and 685 grains/cm$^3$) and Other non-coprophilous fungal spores (mean 80% and 2827 grains/cm$^3$) in Zone G1, however there were two noteworthy peaks in Monolete spores in ca. 6925 BP (80 cm) and ca. 10 365 BP (132 cm).

In Sub-zone G2a (ca. 4495-2885 BP; 53-30 cm), the average of aggregated abundance of non-coprophilous spores did not show much variation in the percentage spore data (mean 95.8%), there was a significant increase shown in the concentration data (17 089 grains/cm$^3$) compared to the other sub-zones. The trend increased steeply until ca. 3600 BP (40 cm) before decreasing in ca. 3320 BP (36 cm) and increasing again to its highest relative abundance (26 292 grains/cm$^3$) in ca. 3175 BP (34 cm), before the G2a/G2b transition. During this sub-zone, a significant peak was observed in the Monolete spores (773 grains/cm$^3$) in ca. 3880 BP (44 cm), Trilete spores (8699 grains/cm$^3$) in ca. 3740 BP (42 cm) and other non-coprophilous spores (22 425grains/cm$^3$) in ca. 3175 BP (34 cm).

In Sub-zone G2b (ca. 2885-1610 BP; 30-4.5 cm), the average of aggregated abundance of non-coprophilous spores was relatively lower on average for the concentration spore data (7359 grains/cm$^3$) but higher in the percentage data (mean 97.4%), likely a result of the Fagerlind effect. The trend in concentration data showed slight fluctuation as it decreased during the sub-zone. Trilete spores were a lot less abundant in both datatypes (mean 7% and 483 grains/cm$^3$) compared to the underlying sub-zone. A decrease in other non-coprophilous spores (mean 6719 grains/cm$^3$) and Monolete spores (mean 157 grains/cm$^3$) was evident in the concentration data whereas the percentage showed an increase in other non-coprophilous spores (mean 88%) and Monolete spores (mean 2%).

In Sub-zone G2c (ca. 1610-1030 BP; 4.5-0 cm), the average of aggregated abundance of non-coprophilous spores was the lowest on average in concentration data (mean 3265 grains/cm$^3$). Monolete spores being at its second lowest relative abundance (mean 107 grains/cm$^3$) and at its lowest (mean 2411 grains/cm$^3$). Trilete spores were on average its second highest relative abundance (mean 22% and 748 grains/cm$^3$) during G2c.

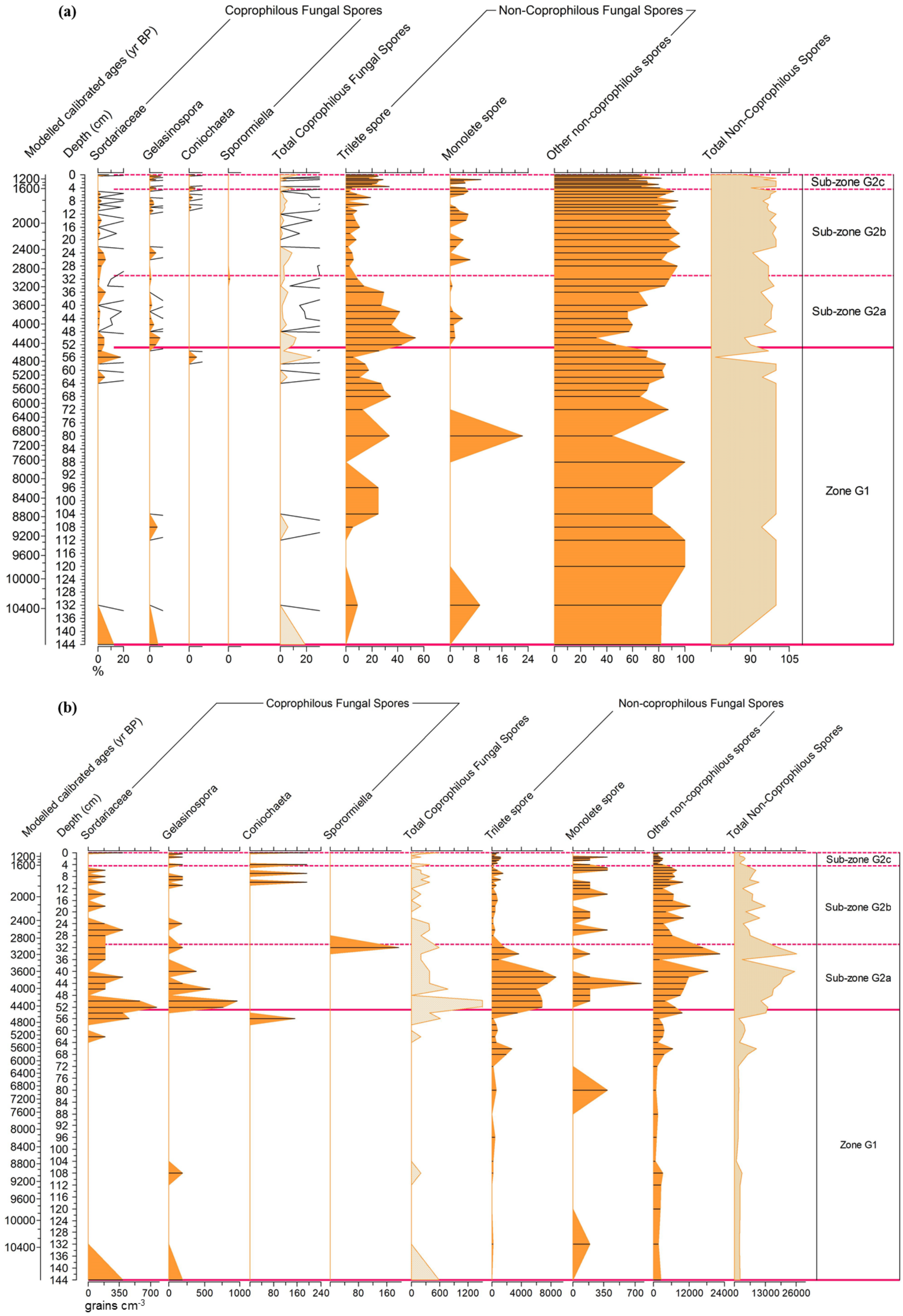



*Figure 5.6: (a) Percentage and (b) concentration spores diagrams for the GWWA006 core at GWWA. Spore types are grouped into coprophilous fungal spores (a proxy for herbivory) and non-coprophilous fungal spores. Statistically significant pollen zones are indicated by the solid line (Zone G1 and Zone G2) whereas dashed lines show sub-zones (G2a, G2b, G2c) and modelled calibrated ages are shown on the far left.*

### 5.5.4 Geochemical (XRF) and physical properties: (Magnetic Susceptibility (MS) and Loss on Ignition (LoI)

In Zone G1 (ca. 10 765-4495 BP; 144-53 cm), geochemical analyses showed a similar trend for Ba, Sr:Ca, Zr:Rb, K and Zn that increased gradually until decreasing during the middle of Zone G1 (ca. 7659 BP (88 cm)) before it increased again (to an unprecedented high for Ba – $2.0E^{-2}$ % in this record) in ca. 5630 BP (66 cm) and then decreased again during ca. 5075-4700 BP (60-56 cm) before increasing again before the G1/G2a transition. Other elements such P was stable at a low level whereas Rb:K (salinity ratio), Mn, Fe and Cu were mostly absent throughout Zone G1. Cs was undetectable often in G1 with a slight increase between ca. 8740-7660 BP (104-88 cm) and another fluctuating trend between ca. 4885-4600 BP (58-54 cm), just before the G1/G2a transition. Magnetic susceptibility (MS) was relatively stable (mean $1.1E^{-7}$ SI) throughout Zone G1 with a slight decrease in ca. 7659 BP (88 cm) and slight increase in ca. 4698 BP (56 cm) before the G1/G2a transition. On average, $LoI_{550}$ (organic carbon) remained stable at a low level in Zone G1.

In Sub-zone G2a (ca. 4495-2885 BP; 53-30 cm), geochemical analysis continued to be stable for Zr:Rb, P and Zn whereas elements Ba, Sr:Ca, K and Cs showed a lot more variability within sub-zone G2a. Noteworthy co-occurring peaks are seen in ca. 4290-4425 BP (50-52 cm) and ca. 3880 BP (44 cm) and troughs in ca. 4160 BP (48 cm) and ca. 3600-3320 BP (40-36 cm). Just before the G2a/G2b transition, noteworthy peaks co-occurred in Sr:Ca and K in ca. 3175 BP (34 cm). Rb:K (salinity ratio), Mn, Fe and Cu remained low throughout Sub-zone G2a. MS continued at a relatively stable trend throughout Sub-zone G2a while on average increasing very slightly (mean $1.1E^{-6}$ SI) compared to the previous zone. The trend in $LoI_{550}$ increased initially, decreased before it increased to an unprecedented peak of 58% in ca. 3027 BP (32 cm) before the G2a/G2b sub-zone transition in the record from this site.

Sub-zone G2b (ca. 2885-1610 BP; 30-4.5 cm) is marked with extreme variability in all eleven geochemical proxies with inverse patterns occurring. When P, Mn, Fe, Cu and Zn were high, Ba, Sr:Ca Zr:Rb, K and Cs were low and vice versa. Sr:Ca was unprecedently low (mean ratio 0.18) compared to previous sub-zones G1 (mean ratio 0.83) and G2a (mean ratio 0.69) in the time period in the sediment sequence from this site. Interestingly, MS experienced an unprecedented peak ($4.8E^{-5}$ SI) just after the G2a/G2b transition in ca. 2745 BP (28 cm) and returned to a stable low trend despite the slightly increased average (mean $3.6E^{-6}$ SI) compared to the previous sub-zones. The peak in MS co-occurred with high relative abundance of Zr:Rb and K. The trend in $LoI_{550}$ decreased, fluctuated at a moderate level throughout the sub-zone.

In Sub-zone G2c (ca. 1610-1030 BP; 4.5-0 cm), extreme variability in geochemical proxies continued with the same inverse patterns. Noteworthy peaks in P, Mn, Fe, Cu and Zn coincided with troughs in Ba, Sr:Ca, Zr:Rb, K and Cs in ca. 1290 BP (1.5-2.5 cm). Sr:Ca decreased even further (mean ratio 0.03) and unprecedented high relative abundances were observed in P ($1.2E^{-2}$ %) in ca. 1440 BP (3 cm), and Zr:Rb (ratio 56.6) and Cs ($5.8E^{-3}$ %) in ca. 1030 BP (0-0.5 cm). MS remained at stable low level (mean $9.6E^{-7}$ SI) and $LoI_{550}$ remained at a moderate level) throughout Sub-zone G2c.

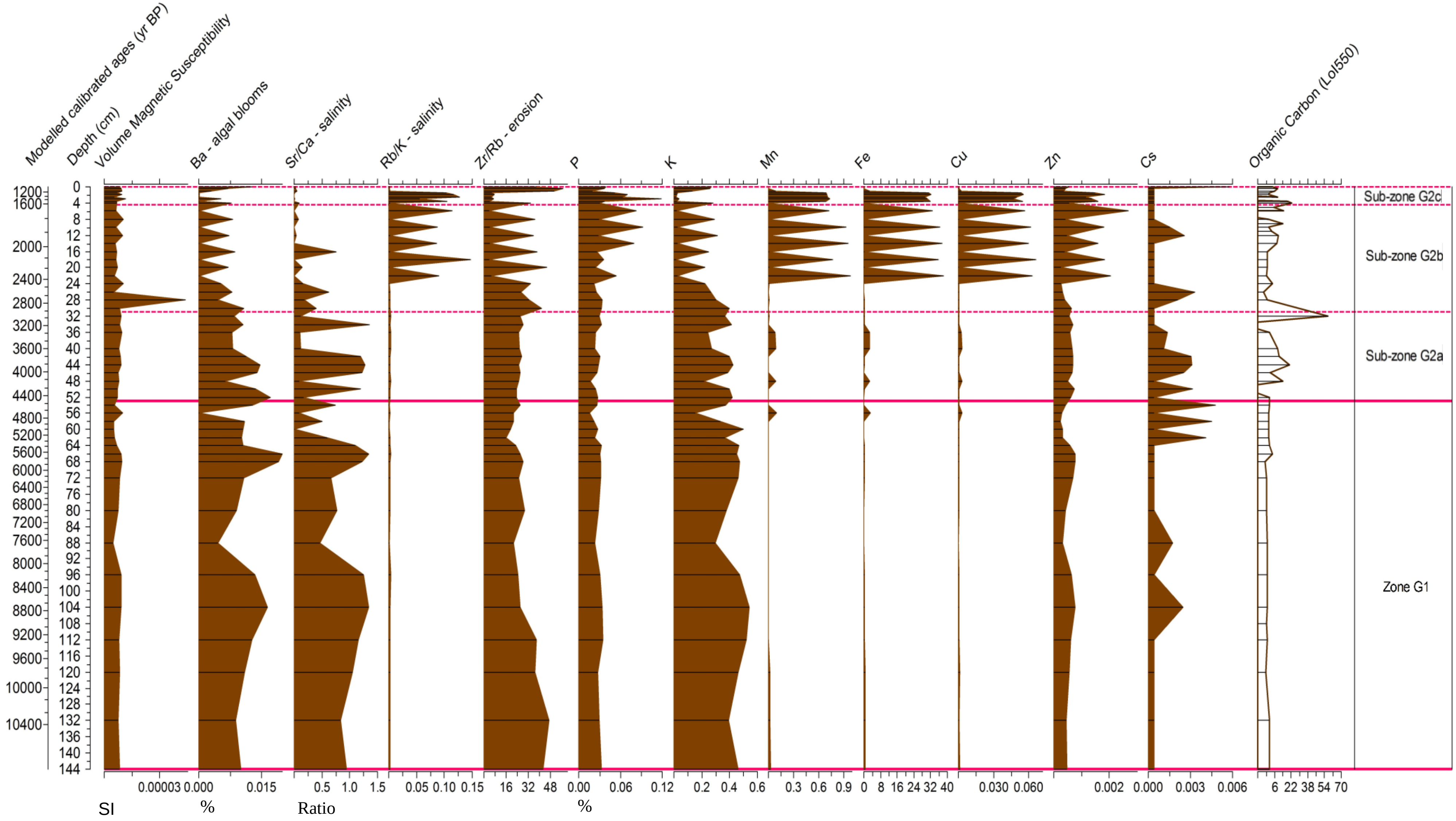



*Figure 5.7: Physical properties (Magnetic Susceptibility), geochemical (elemental composition) and percentage organic carbon (LoI550) content per dry weight for the GWWA006 core as a proxy for soil regulation at GWWA. Pollen zonation depicted on the right and modelled calibrated ages are shown on the far left.*

#### 5.5.5 GWWA006 NMDS ordination results for selected palaeoenvironmental proxies

The NMDS ordination results illustrates the relationship between pollen percentage data and selected palaeo-proxies as explanatory variables (Figure 5.8). The stress value was 0.12 (Figure 10.10) and the stress plot depicted minimal scatter around the line (Figure 10.11) with the original dissimilarities well preserved in the number of dimensions (linear fit $R^2 = 0.89$). The ordination is of satisfactory quality and the similarity relationships between the depths were represented accurately. Pollen zonation analysis (Table 5.4) determined that two statistically different zones: depths 53-144 cm of Zone G1 clustered together whilst depths 0-53 cm of Zone G2 clustered together. These two major groups emerge from a bi-plot of the first and second dimensions (Axis 1 and Axis 2, respectively (Figure 5.8). The projection of pollen types and sample depths onto vectors showed maximum correlation with corresponding environmental variables. The grouping of Zone G1 was associated with an increase in Podocarpus pollen and the soil regulation indicators (Ba – algal blooms, Sr:Ca – salinity, Zr:Rb – erosion and K) gradient as well as an increase in the abundance of *Cliffortia* and degraded pollen. Zone G1 was also associated increased micro-charcoal, Poaceae and Restionaceae pollen. Other explanatory variables associated with lower depths included: macro-charcoal and Magnetic Susceptibility (MS). In contrast, grouping of upper depths (Zone G2) showed that the abundance of *Maytenus oleoides*, Oleaceae, Asteraceae, *Agathosma* and *Anthospermum* pollen increased as the abundance of herbivory (coprophilous fungal spores), P and aquatics and marginal:terrestrial pollen increased.

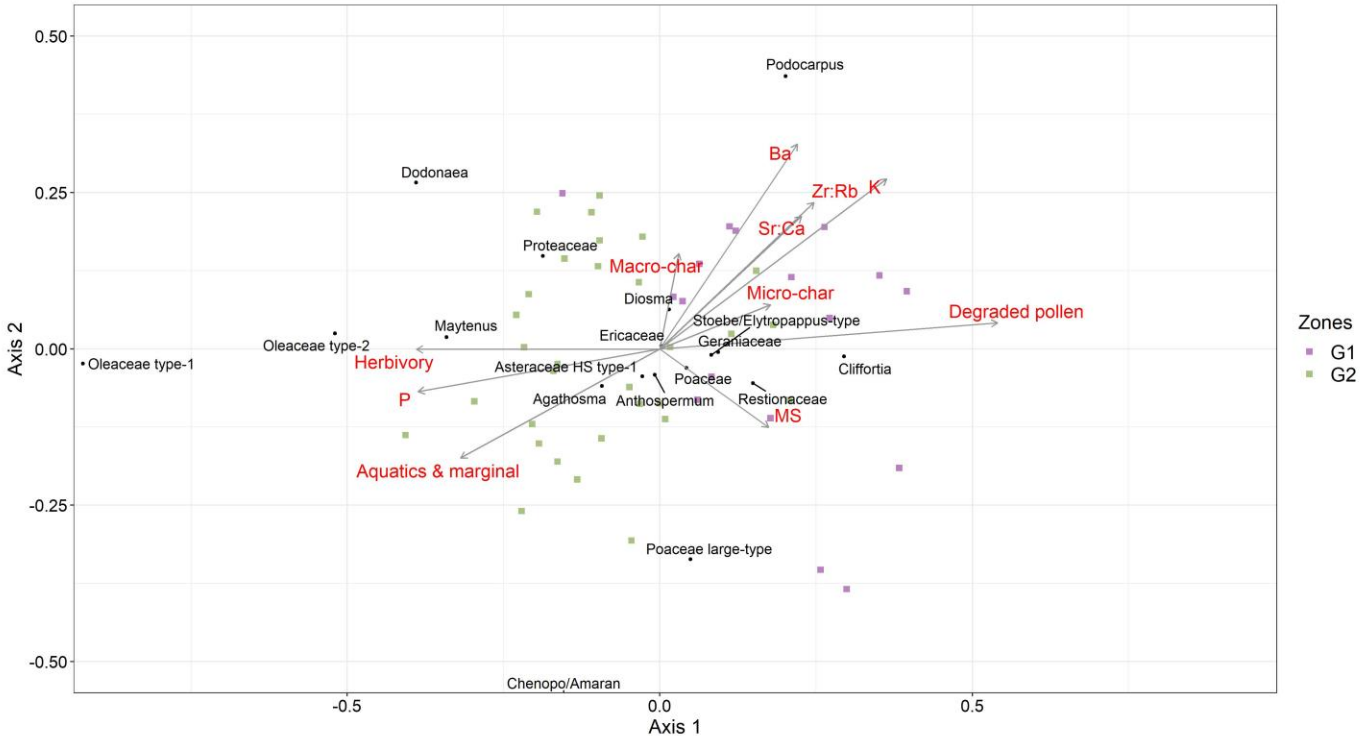



***Figure 5.8: NMDS ordination showing distribution of pollen taxa based on depth for the GWWA006 core. The depth factor was classified into two different zones: G1 (purple) and G2 (green). Ordination based on square root transformed and Bray-Curtis similarity data with a stress factor of 0.12***

## 5.6 Discussion: Using palaeo-proxies to understand the changes in ecological dynamics at GWWA

Synthesis of selected multi-proxy palaeoenvironmental data from GWWA006 core is summarised in Figure 5.9 and patterns in palaeo-proxies and landscape processes are explained. Figure 5.10 shows the multiple palaeo-proxies from the ca. 10 765-1030 GWWA006-record grouped into ecosystem services, including plant biodiversity and soil regulation. Figure 5.11 shows cumulative patterns of changes pollen representing changes in vegetation composition of Winterhoek Sandstone Fynbos. In that section, changes in ecosystem services are interpreted in terms of resilience.

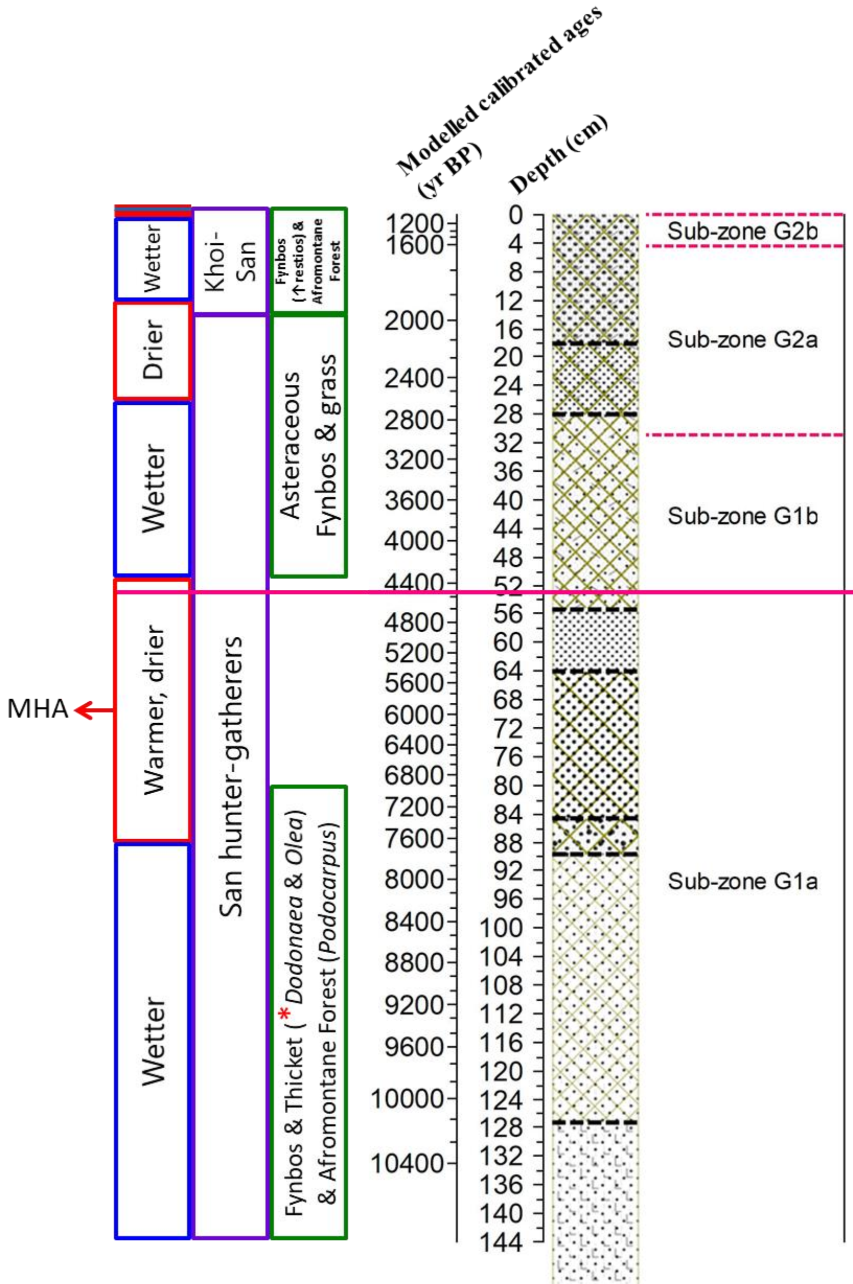
### 5.6.1 Reconstruction and interpretation of environmental history over centennial-millennial timescales

#### *5.6.1.1 Structural Resilience Zone - Drought-Resistant Fynbos Shrubland during warmer, drier conditions in ca. 10 765-4495 BP (Zone G1)*

The results show that the landscape during ca. 10 765-4495 BP indicate a mosaic landscape, with Fynbos vegetation being the dominant patch type, with patches of Thicket/Forest, probably confined to rocky outcrops (section 3.2.1.1) within the Winterhoek Sandstone Fynbos landscape. From 10 765-7600 BP, the Fynbos patches were dominated by *Cliffortia*, with grass co-dominance and Ericas, Restios and Asteraceae also present (see section 5.5.1.2). This mesic Fynbos vegetation (restioid-proteoid-ericoid) with increased *Podocarpus*, indicative of a Thicket/Forest patch (Table 5.2), is consistent with a wetter climate in the early-Holocene. The NMDS plot (Figure 5.8) shows the increased abundance of Cliffortia, Podocarpus, Poaceae and Restionaceae pollen during this time This finding is supported by interpretations of regional palaeo-records (Table 5.1), particularly the mesic Fynbos-Thicket mosaic (Parkington et al., 2000; Scott and Woodborne, 2007a, 2007b) with increased elements of Cape Thicket and Afromontane Forest taxa including Oleaceae spp., *Heeria argentea*, *Maytenus oleoides*, *Cassine peragua*, *Diospyros glabra*, *Dodonaea angustijolia* ca. 10 000-8000 BP (Parkington et al., 2000) and *Podocarpus* ca. 7000 yr BP (Martin, 1968).

In ca. 7600-4495yr BP (88-53 cm) the overall abundance of Fynbos did not change, but there was an increase in more drought-resistant Fynbos vegetation (*Cliffortia*-dominated with increased Asteraceae, Geraniaceae, *Anthospermum* and *Diosma*). This internal reorganisation towards a more drought adapted Fynbos is consistent with other palaeoecological studies in the region (Figure 2.2) that reported climate variability from the early-Holocene to mid-Holocene with a distinct trend of regional aridity during the Mid-Holocene Altithermal (MHA), which also impacted many biomes on the African continent (see section 2.2.2.2) (Chase et al., 2015; Meadows and Baxter, 1999; Neumann et al., 2011; Roberts et al., 2001).

Furthermore, low or no coprophilous fungal spores indicate that herbivory was minimal or absent at the GWWA site in ca. 10 765-4495 BP (Figure 5.9). During this time the fire regime generally varied with moderate local and regional fire levels (mean 325 n/cm$^3$ macro- 13.8 cm$^2$/cm$^3$ micro-charcoal abundance) (see section 5.5.2). Regardless of the climate variability during the early- to mid-Holocene, internal turnover and species reshuffling within the Fynbos community likely provided ecological character resilience to environmental change and therefore persistence of the Fynbos vegetation state. However, the GWWA006 palaeo-record shows increased variability ca. 6180-5805 yr BP (72-68 cm) (during the later part of Zone G1; see Figure 5.10) with multiple palaeo-proxies displaying unprecedented trends in the time period in the sediment sequence from this site, evidence potentially indicative of ecological threshold behaviour, namely: a peak in Fynbos pollen types, including the unprecedented increase in grass pollen coupled with peaks in macro- and micro-charcoal indicative of a simultaneous increase in both local and regional fire. The increased fires were likely caused by hotter and drier temperatures associated with the MHA in ca. 7700-4400 BP (Chase et al., 2015; Neumann et al., 2011). This suggests that climate change (not land-use) was the main driver for crossing an ecological threshold which was not critical due to internal reorganisation maintaining the Fynbos vegetation composition.




**Increasing Fire and Erosion Zone - Grassy Drought-Resistant Fynbos Shrubland**

- Grass-dominated with *Cliffortia* co-dominance. Patch-mosaic of Asteraceae, Ericas and Restios.
- Soil regulation variability with noteworthy erosion event (magnetic susceptibility & geochemical data).
- Warmer and drier climate (inferred from non-coprophilous spores).
- Low herbivory (coprophilous fungal spores), increasing local fire and lower regional fire (macro- & micro-charcoal abundance).

**Transitional Variability Zone - Grassy Asteraceous Fynbos Shrubland**

- Grass-dominated with Asteraceae co-dominance. Patch-mosaic of Restios, Ericas and *Cliffortia*.
- Soil regulation extremely variable (magnetic susceptibility & geochemical data).
- Climate variability - wetting and drying conditions (inferred from aquatics & marginal:terrestrial pollen).
- Low herbivory (coprophilous fungal spores), less local fire and moderate regional fire (macro- & micro-charcoal abundance).

**Within-Biome Scale Resilience Zone - Grassy Ericoid Fynbos Shrubland**

- Grass-dominated with Erica co-dominance. Patch-mosaic of *Cliffortia*, Asteraceae, Restios with Thicket/Forest expansion.
- Soil regulation showed moderate erosion (magnetic susceptibility & geochemical data).
- Wetter climate conditions (inferred from non-coprophilous spores and aquatics & marginal:terrestrial pollen).
- More herbivory, local and regional fire (coprophilous fungal spores, macro- & micro-charcoal abundance).

**Structural Resilience Zone - Drought-Resistant Fynbos Shrubland**

- *Cliffortia*-dominated with grass co-dominance. Patch-mosaic of Ericas, Restios and Asteraceae
- Soil regulation showed moderate erosion (magnetic susceptibility & geochemical data).
- Dry climate (inferred from non-coprophilous spores, degraded pollen and aquatics & marginal:terrestrial pollen).
- Very low herbivory (coprophilous fungal spores), moderate local and regional fire (macro- and micro-charcoal abundance).

***Figure 5.9: GWWA006 schematic summarising how the main findings relate to long-term climatic and land-use change shown on the far left of the diagram.***

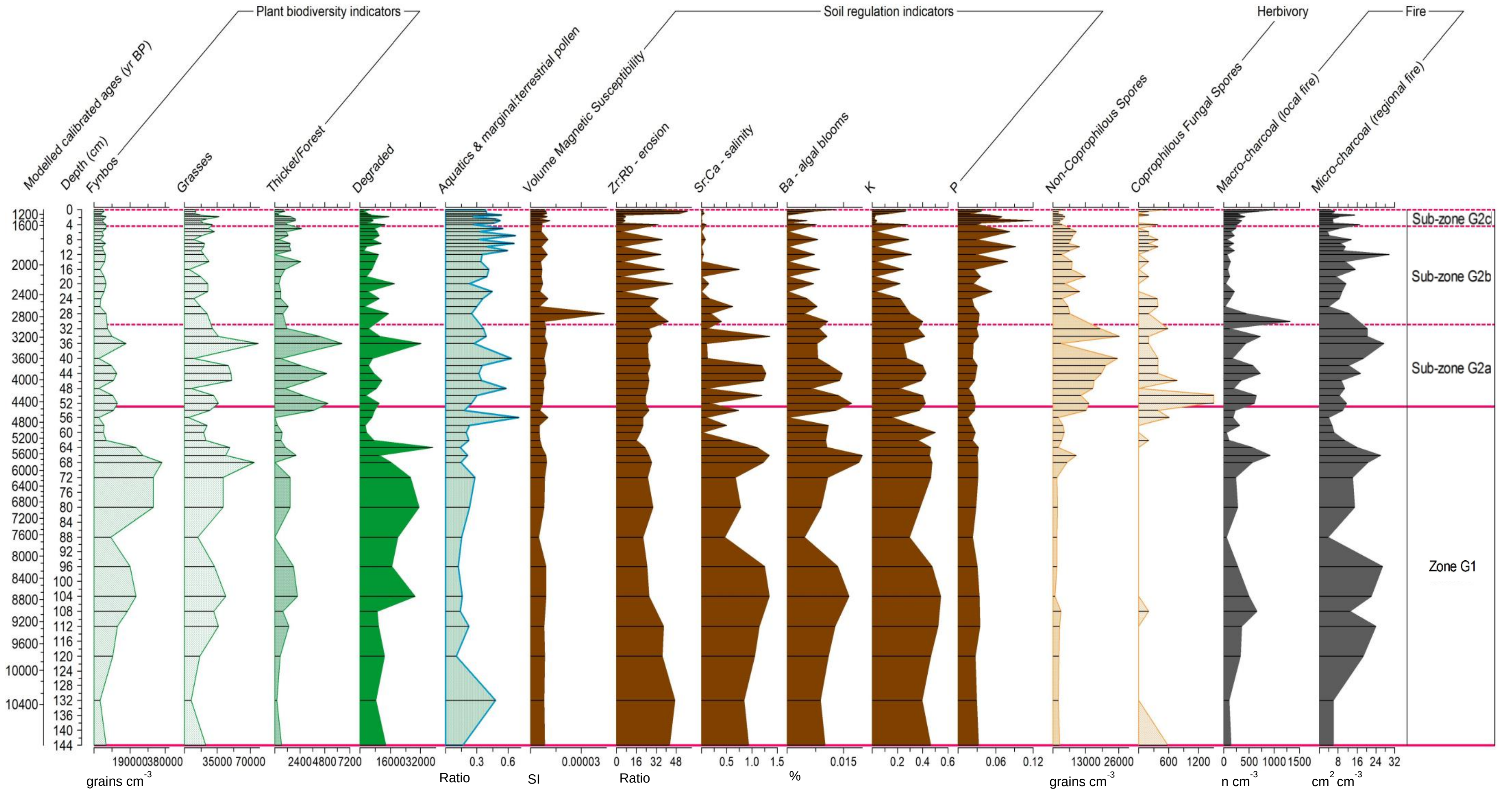



*Figure 5.10: GWWA006 synthesis diagram of multiple palaeo-proxies to extend the timescale of ecosystem services (pollen and geochemical and physical properties) and drivers of change (coprophilous fungal spores and charcoal). Solid pink line indicates statistically significant pollen zonation.*

Prior to a major change in vegetation at the end of Zone G1 in ca. 4495 BP, there are further changes in other indicators that suggest the approach of an ecological threshold and the internal reorganisation in the vegetation structure. For example, there is a change in lithology, which show sediment that contained more fine sand with traces of rootlets (Table 5.3) in ca. 5445-4630 (64-55 cm). There was also evidence of increased erosion, for example, a peak in degraded pollen in ca. 5447 BP (64 cm) possibly related to increasing fire, plus evidence of enriched nutrient water statues indicated by unprecedented high relative abundance of algal blooms (Ba) and salinity (Sr:Ca). Until this point, the *Cliffortia*-dominated drought-resistant vegetation, moderate soil erosion regulation, moderate local and regional fires and low grazing pressure by LIHs (Figure 5.9 and Figure 5.10) buffered environmental changes and provided ecological character resilience.

#### *5.6.1.2 Within-biome scale resilience - Grassy Ericoid Fynbos Shrubland during cooler, wetter conditions in during ca. 4495-2885 BP (Sub-zone G2a)*

The start of Sub-zone G2a at ca. 4495 BP (53 cm) is marked by a major internal shift in vegetation composition as Fynbos became less shrubby and more grassy. This transition can be seen clearly in the NMDS plot (Figure 5.8), which shows a distinct shift in vegetation composition associated with the transition from Zone G1 to Sub-zone G2a occurring at ca. 4495 BP. Overall, these changes suggest a patch-mosaic landscape, where the Fynbos vegetation is dominated by grasses and Ericaceae. Although there was a notable decline in relative Cliffortia pollen abundance, this drought-resistant shrub remained in the landscape together with Asteraceae and Restionaceae, and increased Thicket/Forest patches (Figure 5.9 and Figure 5.10). The increase in rainfall likely gave the opportunity for vegetation to recover including the expansion of Thicket/Forest patches where rocky outcrops act as fire refugia (Figure 6.1 and Appendix Figure 10.5). This is consistent with a wetter and more humid climate, as observed in the regional palaeo-records (Table 5.1) (Meadows and Baxter, 2001; Neumann et al., 2011; Parkington et al., 2000; Scott and Woodborne, 2007a), that favoured a drought-sensitive ericoid community (Skelton et al., 2013; West et al., 2012) with more woody Thicket/Forest-type elements due to increased moisture availability. An increase in the ratio of aquatics and marginal:terrestrial pollen and non-coprophilous fungal spores (see section 5.5.3.2) is also consistent with increased moisture availability.

Therefore, two distinct Fynbos vegetation community sub-types are evident pre- and post- ca. 4495 yr BP (Zone G1/Sub-zone G2a transition) as a result of internal reorganization (decreased *Cliffortia* and increased grasses) that maintained the Winterhoek Sandstone Fynbos vegetation type (Figure 3.2). Alongside these changes in fynbos composition, Thicket/Forest patches expanded. Therefore, signifying within-biome scale resilience despite climate change and increased grazing. Despite this internal reorganisation, fire-prone Fynbos vegetation remained dominant, though in a different grassier sub-type (Figure 5.9 and Figure 5.10), this suggests a quasi-regime shift to a quasi-stable state at the extra-local to regional scale (Figure 5.1) (see further explanation in section 5.6.2 below). This finding is consistent with other examples of internal reorganisation as an important buffering mechanism that enhances Fynbos ecological character resilience to climate change (Gillson et al., 2020).

Alongside the shift in vegetation composition, coprophilous fungal spore results (Figure 5.3 and Figure 5.4) on average showed an unprecedented high (mean 4.2% and 599 grains/$cm^3$) indicating a period of high herbivory at the GWWA site during ca. 4495-2885 BP (53-30 cm) likely linked to wetter climate indirectly supporting an increase in grazing by large indigenous herbivores (LIHs). In turn, feedbacks between herbivory and grass abundance maintains a state dominated by grass (Radloff et al., 2014) (also see Figure 7.9 in Chapter 7 below). In terms of resilience theory, I argue that the new quasi-stable state, Grassy Ericoid-Fynbos Shrubland and Thicket/Forest patch-mosaic, during ca. 4495-2885 BP remains in a less resilient state for ca. 16 centuries (Table 5.4, Figure 5.9 and Figure 5.10). Despite the ecological character resilience, the increased potential of this new state for significant widespread local fires and grazing, which as a result leads to another subtle shift in ecosystem dynamics (see section 5.6.1.3 below) is governed by transitional variability such as another increase in aridity since ca. 3000 BP (Meadows et al., 2010; Neumann et al., 2011). However, since fossil diatoms were not preserved in the GWWA006 sediment core, this study does not showcase an independent climate proxy to verify claims that climate change is the main driver for vegetation change. Therefore, this study does not exclude the possibility of relatively light land-use disturbance during ca. 4495-2885 BP, particularly the manipulation of fire by San hunter-gatherers to stimulate growth of edible plants or using fire-stick farming to attract LIHs to young veld to make hunting easier (Deacon, 1992). The manipulation of fire-grazing dynamics by humans may explain the variability in macro-charcoal, grass pollen and Thicket/Forest pollen abundance during this time.

#### 5.6.1.3 *Transitional Variability Zone - Grassy Asteraceous Fynbos Shrubland in ca. 2885-1610 BP (Zone G2b)*

The pollen record continues to indicate a patch-mosaic landscape comprising of mostly Fynbos patches with Thicket/Forest patches contracting again. Fynbos patches included increased grasses, Restios and *Cliffortia*, however, as Ericaceae declined Asteraceae increased (Figure 5.3, Figure 5.4 and Figure 5.9). The transition from Grassy Ericoid Fynbos Shrubland to Grassy Asteraceous Fynbos Shrubland at the GWWA site in ca. 2885 BP was likely triggered by a shift back to a drier regional climate from ca. 2600-1900 BP (Neumann et al., 2011; Stager et al., 2012)and the combination of increased fuel load from grasses. This resulted in increased local fire evident by an unprecedented peak in macro-charcoal (1311 particles/cm$^3$) in ca. 2890 BP (30 cm) (Figure 5.10). Thicket/Forest abundance declined probably due to their adaptation to higher plant moisture availability.

Therefore, the pollen zonation results showing a state between ca. 2885-1030 BP coinciding with increased macro-charcoal at the start of Sub-zone G2b is indicative of local fire driving the change in Fynbos vegetation sub-type in the GWWA landscape. This finding is supported by the subsequent peak in magnetic susceptibility (MS) thus is consistent with increased erosion and an influx of sediment (Figure 5.10). Geochemical results after ca. 2885 BP show a saw-toothed pattern that is also evident in aquatics and marginal:terrestrial pollen since ca. 1816 BP. These results suggest variability in wetting and drying at the wetland. Despite this variability in climate and local hydrology (De Villiers, 2007) (section 3.1.1), there are no major shifts in vegetation types suggesting that climate variability was still within the Fynbos vegetations' tolerance.

Furthermore, the GWWA006 record suggests that local land-use disturbance by Khoi-San hunter/herders was minimal in ca. 2885-1610 BP (Sub-zone G2b) since the relative abundance of coprophilous fungal spores was low. However, there are at least fifteen recorded Khoi-San hunter/herders rock art sites located within the GWWA (Table 5.1), one of which is located south-east of the GWW006 coring site (see light brown point labelled historical site in Figure 3.2). Dating of the rock art sites at GWWA suggests Khoi-San hunter/herders were in the region up to 100 years ago (ca. AD 1900s) (CapeNature, 2016). Interestingly, the GWWA006 palaeo-record shows evidence of an unprecedented peak in micro-charcoal (29.8 cm$^2$/cm$^3$) in ca. 1890 BP (12 cm) indicative of extensive regional fires during the period when Khoikhoi pastoralists introduced livestock to the Western Cape (Deacon, 1992; Humphreys et al., 1998) (see section 3.1.2). This finding is consistent with Meadows and Baxter (2001)'s report of decreased Poaceae pollen and increased micro-charcoal attributed to Khoikhoi pastoralists and/or climate change since 1900 BP at Klaarfontein Spring (Figure 2.2a).

Despite the regional climate variability (Neumann et al., 2011; Stager et al., 2012) and highly variable patterns in soil regulation, the grassy-shrubland state that was dominated by Asteraceous shrubs during this transitional period was maintained by low levels of herbivory and an unprecedently low local fire regime, similar to pre-colonial landscape patterns reported by Forbes et al., (2018) at Elandsberg PNR (Figure 2.2a) in ca. AD 750-1950.

#### 5.6.1.4 Increasing Fire and Erosion Zone - Grassy Drought-Resistant Fynbos Shrubland in ca. 1610-1030 BP (Sub-zone G2c)

The patch-mosaic landscape mainly comprising Fynbos and Thicket/Forest patches was maintained. The grassy-shrubland component of the landscape still included grasses, Ericas and Restios, however, Asteraceous shrubs decreased while *Cliffortia* increased again during ca. 1610-1030 BP. A hotter and drier climate associated with the later Medieval Climate Anomaly (MCA) (ca. AD 900–1400 or 1050-550 BP) (Lee-Thorp et al., 2001; Nicholson et al., 2013; Stager et al., 2012) likely could have influenced the recovery in drought-resistant taxa such as the increasing trend of relative abundance of *Cliffortia* pollen seen in the percentage data abundance in ca. 1030 BP. Interestingly, in both warmer periods of the MHA and MCA, *Cliffortia* pollen abundance increased (Figure 5.3). Holden et al., (2021) reported that some landowners felt that recent increases in specific indigenous species such as *Stoebe plumosa/Seriphium plumosum* and *Cliffortia* spp were especially problematic as the dense vegetation causes large and hot wildfires. This is supported by the palaeo-record showing an increase in *Cliffortia* pollen abundance coupled with an increase in macro-charcoal between 1610-1030 BP (Figure 5.3), suggesting an increased local fire regime at the site. Therefore, increasing *Cliffortia* might be an expected and appropriate response to warming climate and increased fire occurrence in the last 40 years. The modern vegetation survey showed two *Cliffortia* spp, *Cliffortia ruschifolia* and another unidentifiable *Cliffortia* spp (Table 5.2) but these two types were not distinguishable in the pollen record. Additional research to improve the resolution of *Cliffortia* pollen identification (e.g. Macpherson, 2016) may be useful for determining whether it is a disturbance indicator or a drought/warming adaptation indicator for ecosystem resilience.

Similarly to the reported increase in fires at GWWA since ca. AD 1970s (Holden, 2017; Holden et al., 2021) the increased macro-charcoal in ca. 1030 BP is not unprecedented within the ca. 10 765-1030 charcoal record as an even higher relative abundance of macro-charcoal is evident at the start of the transitional period in ca. 2885 BP. Moreover, results from the site suggests that during ca. 1610-1030

BP, a warmer climate associated with the MCA possibly impacted on not only an increased local fire regime but also soil dynamics. Geochemical results show erosion variability with unprecedentedly high erosion (Zr:Rb), P and Cs (Figure 5.10) relative to the time before this.

However, since the pollen zonation results (section 5.5.1.1) are not significant at the G2b/G2c sub-zone level, results suggest that plant biodiversity as a provisioning and regulating ecosystem services responded negligibly to climate change during this period compared to the impacts of climate variability from the hot and dry MHA to wetter conditions after ca. 4495 BP resulting in a quasi-stable state. This finding is similar to the results from the palaeo-record from at Elandsberg PNR which showed no major vegetation changes during the MCA although there was evidence of regional fire peaks (Forbes et al., 2018). However, there was uncertainty whether the increase in regional fires were caused by the hot and dry climate of the MCA or perhaps intense burns by Khoikhoi pastoralists who would have cyclical burns to produce pastor for their livestock.

The wetland characteristics consisted of several dominant taxa (Table 5.2) many of which are represented in the pollen record indicative of changes in wetland dynamics over the ca. 10 765-1030 GWWA006-record in response to climate variability and land-use impacts. Regrettably, without comparative palaeo-data covering the period of potential impacts from European settler agriculture and associated crop and livestock farming since ca. 300 BP (ca. AD 1650), the cause for recent changes in ecosystem services and drivers is not easily decipherable from the present record. The Zuurvlakte Farm dam upstream from the coring site associated with agricultural activities including the establishment of protea cut flowers cultivation and subsequent berry farming would have likely influenced the wetland dynamics and less sediment accumulation in recent decades. Agricultural activities since AD 1909 when settlements were built near De Tronk (Figure 3.2) increased during the 20$^{th}$ century (Table 5.1) and agricultural intensification could have contributed to periodic scouring by flood water resulting in a negative deposition rate and the lack of consolidated sediment representing the past 1000 years. Additional explanations could be that the top sediment is missing due to natural cessation of accumulation or that sediment was lost during core retrieval and transportation to UCT for storage. We did not but we When determining the chronology, bulk sediment was analysed for AMS radiocarbon dating as this corresponds most closely with the pollen record. However, another consideration in future would be to date macro-charcoal individually to corroborate the dating for the top of the core, and to determine whether sediment has been removed because of burning by modern wildfires. Nevertheless, whether climate or land-use caused recent changes in ecosystem services, such as soil regulation variability and increased *Cliffortia*, the intrinsic resilience of Fynbos continued despite transitional environmental dynamics evident since ca. 2885 BP persisting until ca. 1030 BP. This long-term palaeo-perspective on changes in ecosystem services and drivers provides valuable contextual evidence to consider while developing current and future management targets and strategies for GWWA and the surrounding multi-functional landscape.

### 5.6.2 Resilience theory and boundary crossing

#### *5.6.2.1 Palaeoecological insights linked to ecological thresholds and potential management thresholds at GWWA*

Although the upland conservation site historically experienced less land-use disturbance in comparison to the other case study sites, the GWWA006 palaeo-record showed evidence of internal reorganisation that mitigated changes in climate. Specifically, a shift from a Drought-Resistant *Cliffortia*-dominated Fynbos shrubland and Thicket/Forest patch-mosaic to a Grassier Mesic Fynbos and Thicket/Forest patch-mosaic was coupled with changes in soil regulation, local and regional fires and subsequent increased herbivory (Figure 5.8 and Figure 5.10). This state shift was driven by climate change in ca. 4495 BP from the hot and dry MHA (ca. 4400-7700 BP) to a wetter and humid climate (ca. 2000-4000 BP). The shift in in plant biodiversity was likely due to non-linearity in vegetation-fire dynamics in ca. 6180-5805 yr BP impacting on soil erosion regulation in ca. 5445-4630 BP before crossing an ecological threshold in ca. 4495 BP (section 5.6.1.1).

Changes in fossil pollen results show internal reorganisation within the Fynbos vegetation type indicated by a statistically significant transition from one quasi-stable state (Zone G1) to another quasi-stable state (Zone G2a, b and c). States are regarded as quasi-stable in that they remain stable for a significant length of time without crossing a critical ecological threshold (Jackson and Wood, 2018). Since the GWWA006 palaeo-record detected environmental change that did not occur at an ecotone or between-biome scale such as Renosterveld-Fynbos, Succulent Karoo-Fynbos and Fynbos-Forest ecotones in other studies within the CFR (e.g. Forbes et al., 2018; MacPherson et al., 2019, 2018), it is classified as a shift between quasi-stable states of Fynbos vegetation sub-types at the within-biome scale. Thus, changes in sub-types of Winterhoek Sandstone Fynbos (section 3.2.1.1) (Mucina and Rutherford, 2006) signifies ecological character resilience at the within-biome scale despite changes in climate, fire and herbivory.

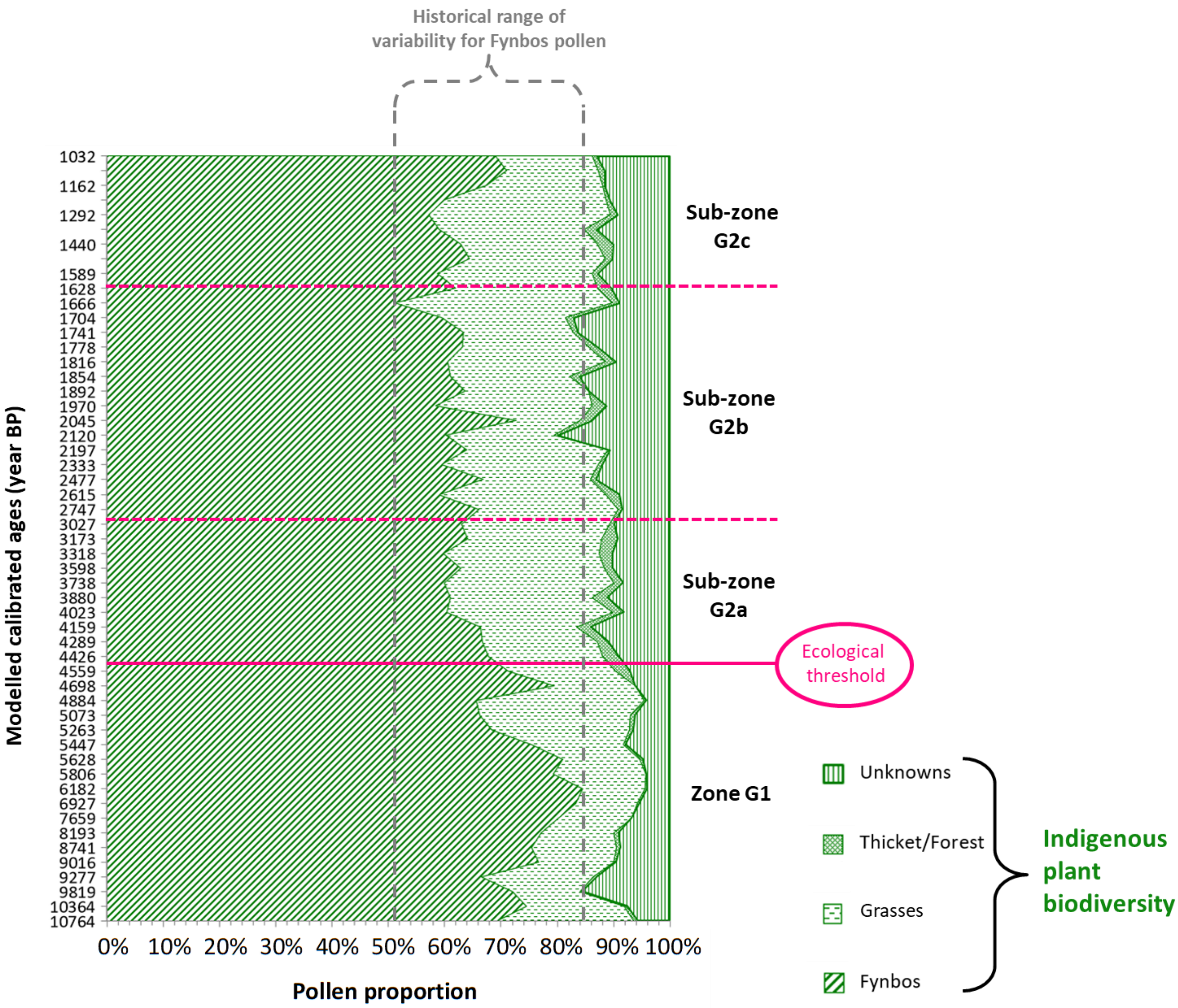



*Figure 5.11: Indigenous plant biodiversity aggregate of fossil pollen data from the GWWA006 core expressed as percentage values. The pink solid horizontal line separating zones G1 and G2 indicates statistically significant zonation and representation of an ecological threshold, while the pink dashed horizontal lines indicate sub-zones. The grey dashed lines represent a historical range of variability for Fynbos pollen, which could provide a basis for discussing management thresholds based on the pollen record with the assumption that the aggregated value of Fynbos shrubland pollen should not decrease below its historically lowest value (51%) in ca. 1665 BP.*

The resilience of Fynbos vegetation is achieved via the emergence of quasi-stable states over centennial-millennial timescales. The diversity of the Fynbos flora provides inbuilt adaptability and functional redundancy (Hunter, 2011; Jia and Whalen, 2020) that buffers this vegetation type from environmental change. The wide variation in altitude and rainfall across the Fynbos biome contributes to the wide physiological tolerance of the biome as a whole (section 2.2.2.1). Even though the past is not a perfect analogue, the process-based understanding of past resilience may be relevant for understanding the mechanisms that confer future resilience. Resilience of Fynbos vegetation in the uplands of the CFR has also been reported by several other palaeoecological and neoecological studies (Meadows et al., 2010; Meadows and Sugden, 1993, 1991a; Quick et al., 2011; Slingsby et al., 2014). In terms of theories of complexity, the resilience is likely due to slow processes (Dearing et al., 2015) of internal reorganisation and turnover of Fynbos taxa controlling the long-term resilience over the last ca. 10 000 years. Abiotic processes, including increased fire and changes in soil regulation during a transition from a warmer and drier to a wetter climate (Table 5.1) resulted in an ecological threshold being crossed (Figure 5.11). This ecological threshold represented a quasi-regime shift at the within-biome scale because Fynbos was still dominant, thus ecological and structural character did not change pre- and post-4495 BP. However, interacting effects of climate and land-use disturbance present confounding factors that make it increasingly difficult to attribute causes when multiple variables, for example, increased fire and erosion coupled with changes in vegetation, are changing at the same time and show non-linearity (Bennion et al., 2011a).

The GWWA006 palaeo-record shows internal reorganisation of vegetation sub-types that would have maintained all of the elements present in the mosaic landscape, but with variation in the composition of Fynbos. The persistence of this mosaic landscape is explained by the wide range of drivers (fire and grazing) and microclimate refugia (e.g. rivers and rocky outcrops) (Oliver et al., 2015). This internal reorganisation is demonstrated through the new quasi-alternative stable state being maintained by feedbacks between increased plant available moisture, fire and biotic processes such as increased grazing and the shuffling of taxa evident via changes in the relative abundance of Thicket/Forest and Grassland pollen types while maintaining Fynbos-dominance throughout the record. All three vegetation sub-types remained present over time. This claim is congruent with an applied palaeoecological study located at a fynbos-forest ecotone at Platbos 1 (Figure 2.2), which showed intrinsic resilience properties via Fynbos species turnover and functional redundancy at the temperate limit of the Fynbos biome (Gillson et al., 2020; MacPherson et al., 2019). In the present study, Restionaceae taxa could be a plant biodiversity indicator at GWWA since the modern vegetation survey indicated an overwhelming restio diversity and abundance with estimated 26% in the Taller Restioid shrubland on the western sandy slope (VU1a) and 50% in the Shorter Restioid shrubland, on the eastern sandy slope (VU1b) (Figure 10.5, Figure 10.6 and Table 10.3). Interestingly, restios were always present in the GWWA006 pollen record such that the internal turnover of Fynbos taxa shows persistence of Restionaceae pollen but never restioid dominance (Figure 5.3, Figure 5.4 and summarised in Figure 5.9). However, future research is needed to determine the mechanistic reason for the ecological character resilience and testing of multiple biodiversity indicators (e.g. *Cliffortia* restios and grasses), which could be conferred by the following resilience criteria: Hardness; Recovery; Robustness; Elasticity; and Index of resilience (Herrera, 2018, 2017) and resistance or negative/positive resilience (Lake, 2013; Oliver et al., 2015).

#### *5.6.2.2 Further research and implications for management decision-support at GWWA*

With the lack of consolidated sediment representing the past 1000 years, it is uncertain whether land-use impacts during European settler agriculture, agricultural intensification, and conservation practices since ca. AD 1985 and the interacting effects of recent climate warming (Table 5.1) have affected ecosystem services provision at the site. However, results from the GWWA006 record can still add to the state of knowledge of the relatively poorly studied region of the Groot Winterhoek Mountains (Mucina and Rutherford, 2006) by extending the temporal timescale and building on transdisciplinary research conducted by Holden, (2017) that showed the convergence of multiple knowledge systems (social science, ecology, environmental geography, geomatics and hydrology) to understand historical ecosystem service trade-offs and how this is perceived by landowners (section 3.4.1). Holden, (2017) reported a 11-30% increase in Fynbos vegetation cover within the GWWA despite increased annual temperatures and decreased summer rainfall and annual wind run over the last ca. 40 years. However, increased vegetation cannot be attributed to GWWA protected area establishment since increased land cover also occurred outside the nature reserve. Changes in plant biodiversity have resulted in increased fuel accumulation and therefore a change in the fire regime including increased fire size, fire return intervals and fire intensity (Holden, 2017; Holden et al., 2021). Landowners perceive recent increased land cover both in the reserve and the surrounds is linked to changes in land-use policy that resulted in increased local fire since ca. AD 1970s with potential impacts on water supply and soil erosion (Holden, 2017; Holden et al., 2021).

As baselines shift in a changing world (Forbes et al., 2018; Gillson, 2015; Manzano et al., 2019; Pauly, 1995; Wolfe et al., 2012), extending the timescale of ecosystem services beyond the 50-year account is valuable. The ca. 10 765-4495 BP GWWA0006 record provides a more contextual evidence-based understanding of change over time that goes beyond our collective societal memory and provides a pre-colonial benchmark for the future (Forbes, 2014). The long-term palaeo-perspective suggests that the recent increase in land cover during the 20$^{th}$ and 21$^{st}$ century reported by Holden (2017) is likely within the historical range of variability. The ecosystem has retained functionality despite climate variability, particularly during the hot and dry MHA and MCA. Therefore, in both warm periods covered by the GWWA0006 palaeo-record, the adaptive capacity of plant biodiversity behaved resiliently through changes in dominance of drought-adapted Fynbos taxa, *Cliffortia*. Such insights are important in refining predictions of how Fynbos might respond to warming/drying climates and provide a basis for generating hypotheses about resilience and management thresholds. For example, the historical range of variability for GWWA Fynbos pollen data, which is 51-84% (Figure 5.11), can be calibrated with quantitative estimates of current Fynbos vegetation cover at GWWA. Discussion of the calibrated historical range of variability would form the basis for deciding TPCs or management thresholds in collaboration with stakeholders as a guideline for the future (Biggs et al., 2011; Biggs and Rogers, 2003; Dearing et al., 2014; Gillson et al., 2021; Gillson and Marchant, 2014; McLoughlin et al., 2011; Rogers, 2003). If it is social-ecologically feasible, a possible management threshold might be to return burning and soil erosion regulation to pre-ca. 4495 BP levels, a time when Fynbos pollen was relatively higher (mean 75%) and grass pollen relatively lower (mean 17%). Therefore, management thresholds would trigger corrective action such as controlling and preventing fires or decreasing intensity of agriculture or upstream river damming.

Although the landscape of the older reference condition, MHA, could be considered as the most ‘pristine’ and thus a more 'desirable scenario', it may not be the most suitable conservation or restoration target given greater biotic and abiotic changes have taken place between the MHA reference condition and the present day (section 2.2.2.2) (Bennion et al., 2011a). Therefore, the system state during the more recent MCA (ca. AD 900–1400), which was a Grassy Drought-Resistant Fynbos shrubland, may be a reasonable and useful past analogue. As the landscape or ecosystem was historically durable, insensitive or resilient achieving such a target may be ecologically possible. Although the pollen record suggests a grassier shrubland, the presence of Fynbos taxa such as drought-sensitive Ericaceae throughout the GWWA006 palaeo-record as well as in the modern vegetation survey (section 5.2) is possibly another indicator of a highly resilient ecosystem despite climate, fire and grazing variability. Such functional redundancy is important in terms of land-use management because the CFR is a spatially and temporally complex, multi-functional landscape and it is essential to consider what the limits of acceptable change are when choosing appropriate reference conditions (e.g. MHA or MCA) and setting realistic management targets for effective integrated catchment management. Adaptive ecosystem-based management with intermediate levels of fire and grazing disturbance and a heterogeneous mosaic landscape allows a greater range of plant species to persist and therefore promote ecosystem services provision (Forbes et al., 2018; Hanski, 1991; Molles, 2015; Peng et al., 2012; Walton, 2006; Wu and Loucks, 1995) (section 2.1.2.2).

Current fire regimes at GWWA may still be within the historical range of variability. Fossil charcoal results indicate unprecedented peaks in local fire (macro-charcoal) and regional fire (micro-charcoal) in ca. 1610-2885 BP (Sub-zone G2b) during possible occupation by indigenous Khoi-San hunter/herders (section 5.5.2 and Figure 5.10). Interestingly, similar macro- and micro-charcoal peaks are also evident in ca. 4495-10 765 BP (Zone G1) when there was limited land-use impact. Furthermore, together the modern vegetation survey results (section 5.2) and aggregated pollen record (see grey dashed vertical lines in Figure 5.11) suggest that plant biodiversity is still within the historical range of variability which could be defined as a palaeo-safe operating space when consulting stakeholders to decide on management thresholds. However, using the past pollen and charcoal records to infer the future is still quite speculative. Therefore, further work could include additional palaeoecological analyses covering the most recent period, to compare how variability in key ecosystem parameters (biodiversity, fire, herbivory, soil erosion regulation and water quality) over the past 300 years compares with the longer-term record.

Given the diversity of *Cliffortia* species, and the ecological importance of *Cliffortia* as a drought adapted taxon in the Fynbos pollen record (MacPherson et al., 2019, 2018), it could be valuable if the pollen morphology could be resolved beyond the genus level. This would allow the ecology of *Cliffortia* to be more accurately interpreted in the pollen record. Furthermore, instead of relying fully on fossil pollen to reconstruct quaternary climates based on known climatic affinities of indicator plant species (e.g. drought-resistant Ericaceae or drought-tolerant *Cliffortia, Stoebe* or Oleaceae), separate palaeo-climate proxies such as diatoms or carbon isotopes from other cores within the GWWA (Figure 2.2) can be used to understand the effect of climate on vegetation. Moreover, the analysis of additional palaeo-records over decadal-millennial timescales from different landscapes within the region (e.g. see Chapter 6 and 7) will assist in better understanding landscape processes and changes in ecosystem services in response to drivers under past climate and land-use change.

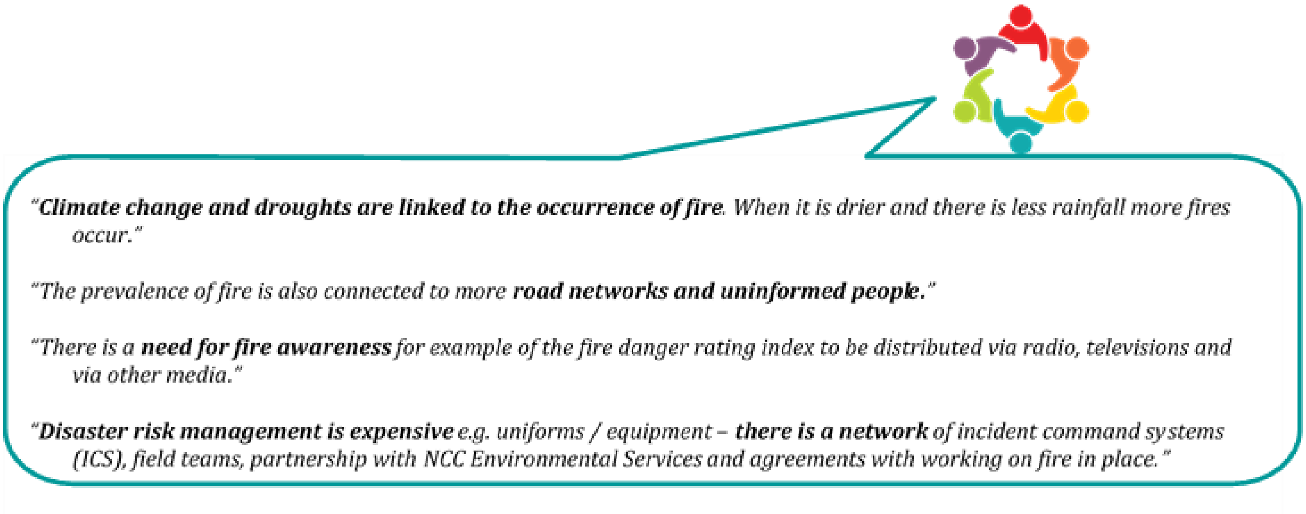



***Figure 5.12: Stakeholder perceptions on fire within the Middle Berg River Catchment of the Cape Floristic Region (CFR).***

However, further quantification of ecosystem services into metrices that are operational to land-use managers is a priority. Thus, the calibration of pollen, charcoal, spores and geochemical markers is needed to situate the historical range of variability with current ecological parameters such as land cover, fire and grazing frequency and intensity and soil erosion (Table 1.2) to support simulation studies and landcover reconstruction in the region. Future pollen trap studies (e.g. Hill and Finch, 2021) would provide an even more robust interpretation of palaeoecological results and inform the next step which is to integrate the long-term records from multiple data sources with stakeholders' perceptions of change. This provides context for the comparing the historical range of variability (e.g. Figure 5.11) with the present landscape, and a basis for setting management thresholds such as TPCs (Biggs et al., 2011; Gillson and Duffin, 2007). Such TPCs could be described as a palaeo-safe and just operating space which could be a value-based and scientifically rigorous guideline for decision-makers. (Dearing et al., 2014; Hossain et al., 2017; Mckay et al., 2016; Verburg et al., 2016) In the present study, for example, land-use managers representing Commercial Farming, Conversation Practice and Governance Authorities shared their understanding on the causes and impacts of fire and the management thereof via semi-structured interviews and the multi-stakeholder engagement workshop. Note that several noteworthy quotes are shown in Figure 5.12 with the detailed systems map illustrated in Figure 10.24, however, the integral role of key stakeholders within the Elandsberg PNR context is discussed later in sections 7.1.2 and 0. With this in place, to advance the contextual evidence, it is recommended that further dynamic- and causal- thinking is applied. An analysis of future SES scenarios with multiple pre-colonial reference conditions can provide a more operational, process-based and prescriptive understanding of resilience. PSD modelling can further test hypotheses about resilience. Future scenarios analysis can better inform TPCs and management targets to maintain current and future ecological integrity at GWWA. Defining and testing management thresholds for fire and grazing regimes that influence plant biodiversity levels, water quality and soil erosion regulation are intended to support decision-makers to systemically improve climate-resilient preparedness to natural disasters.

### 5.6.3 Concluding remarks for GWWA SES

The GWWA006 multi-proxy palaeo-record provided a valuable local to regional signal of changes in ecosystem services such as plant biodiversity and soil erosion regulation and drivers such as climate, fire and grazing during the Holocene. The vegetation state change at GWWA is representative of internal reorganisation buffering environmental change. The disturbance-adapted vegetation enabled Fynbos to persist by internal reorganisation despite climate change and minimal land-use disturbance by indigenous Khoi-San hunter/herders. Results confirm findings from other studies (Forbes et al., 2018; Gillson et al., 2020; MacPherson et al., 2019) that suggest that decreased rainfall during drier periods such as the MHA and MCA favoured fire and Drought-Resistant Fynbos Shrubland persistence dominated by *Cliffortia*, while in wetter periods mesic (restioid-proteoid-ericoid) Fynbos shrubland that was grassier and Thicket/Forest elements increased (Figure 5.10). Internal reorganisation of taxa resulted in a quasi-regime shift in ca. 4495 BP (Zone G1/ Sub-zone G2a transition) between sub-types of Winterhoek Sandstone Fynbos vegetation. Further quantification of ecosystem services and drivers by calibration with modern analogues alongside additional stakeholder engagement with key informants could help to determine whether the pre- or post-4495 yr BP quasi-stable states are representative of the modern vegetation at the GWWA site (Table 5.2 and Figure 5.1). With a highly variable past and an uncertain future, I recommend that land-use managers consider climate-resilient reference conditions/states from the past. Although the MHA or MCA are imperfect analogues for the future, these reference conditions may be useful and reasonable analogues for current and future climate warming and planning for major fire events when defining social-ecologically-sound management and restoration targets.

# 6 CHAPTER 6: STATE CHANGE AT RHENOSTERVLEI FARM: APPLIED PALAEOECOLOGICAL RESULTS AND DISCUSSION

Within the Middle Berg River Catchment, and elsewhere in the Cape Floristic Region (CFR), long-term economic development needs to be considered in the context of ecological degradation and overall social-ecological function, specifically the provision of ecosystem services. Although plant biodiversity is monitored within formal protected areas of the CFR, this is rarely the case in areas that are not formally protected (e.g. on private lands located mostly in the lowland regions). Furthermore, monitoring only began in recent decades when substantial alteration of ecosystems had already taken place. This chapter shows the importance of a multidecadal palaeo-perspectives on changes in the three selected ecosystem services, plant biodiversity, water quality and soil erosion regulation, and drivers of change such as herbivory and fire disturbance and climate. This approach is demonstrated using palaeo-proxy records from a sediment core retrieved from a small wetland on Rhenostervlei Farm covering the period ca. AD 1795-2011, combined with known historical records from the Farm and the general surrounding of the Middle Berg River Catchment. Pollen percentage data were aggregated into vegetation types to understand the relationship between local plant biodiversity compared to exotic taxa. Can the reconstruction of environmental history from multi-proxy, high temporal resolution palaeo-data provide insights on whether this site is ecologically degraded? Can the ca. 215-year-old RV3 palaeo-record from Rhenostervlei Farm provide clues as to whether ecosystem function is compromised, and social-ecological system (SES) resilience is completely lost in these highly transformed lowlands of the CFR? What do the results suggest for restoration - should the site be restored to pre-impact levels?

## 6.1 Historical Timeline of events: Changes in direct and indirect variables

***Table 6.1: Timeline of events describing long-term climatic, vegetation and land-use change (direct and indirect) during the RV3 core period (ca. 215 years). Columns on the far left summarise the land-use types/groups, climate and dominant vegetation (green shading) during the respective time periods. As visual snapshot, the climate column indicates red shading for warmer and drier climates and blue shading for wetter and colder climates. References are provided next to the text for each historical account with text specific to Rhenostervlei Farm highlighted in bold. References in the climate column as specific to the climate.***

| Land-use types/groups | Climate | Dominant vegetation | Variables with direct land-use impacts | Variables with indirect land-use impacts |
|---|---|---|---|---|
| Agricultural intensification (AD 1900-1985) | 20th & 21st century warming<br><br>(Haensler et al., 2011; Hoffman et al., 2011; Meadows, 2006; Turpie et al., 2002) | Shrubby grassland with increased Renosterbos | • **Significant drought conditions. Summer 2018 reportedly the first time in 55 years that there was no water in the Berg River adjacent to the RV3 site. The Water allocation was reduced by 60% and the farmer could only fill up 6/15 waterblommetjie dams (Gildenhuys pers. comm. 2019).**<br>• **Flooding on Rhenostervlei Farm (influx from river) in AD 2007 (Gildenhuys pers. comm. 2019).**<br>• **Further land modification at Rhenostervlei Farm agricultural intensification (clearing with tractor) in AD 1990 (Gildenhuys pers. comm. 2019).**<br>• **Farmer Altus Gildenhuys took ownership of Rhenostervlei Farm, subdivided it into 24 fields (wheat, oats, canola and fodder) and used crop rotation ("ouland"). Cattle graze freely in natural vegetation remnants since AD 1984.** | |
| | | Increase in Asteraceous shrubs, decline in grasses | • **Flooding on Rhenostervlei Farm in AD 1977 (Gildenhuys pers. comm. 2019).**<br>• **Intensification in farming practices from the use of donkeys to farming with heavy machinery in AD 1960 (Gildenhuys pers. comm. 2019).** | • Due to increased water demand for Cape Town (1.7 million cubic metres per day), Voëlvlei Dam wall was raised with more water extracted from Klein Berg River in AD 1969 (River Health Programme, 2004).<br>• Voëlvlei Dam commissioned in AD 1952- water supplied to Riebeek-Kasteel, Riebeek-Wes, Malmesbury, Darling, Moorreesburg and farms along the supply route (River Health Programme, 2004).<br>• Precipitation maxima ca.20 cal BP (ca. AD 1930) (Stager et al. 2012).<br>• Wheat Importation Restrictions (Act No. 10 of 1930); Soil Erosion Adversary Council established (AD 1930) (Meadows 2003; Hoffman and Ashwell 2001) |
| | | Increased Asteraceous shrubs, declining grasses, *Acacia* & *Eucalyptus* introduced | • **Grandfather of current farmer Altus Gildenhuys purchased Rhenostervlei Farm in AD 1906 (Gildenhuys pers. comm. 2019).** | |
| European settler agriculture<br><br>(AD 1650-1900) | 19th C rainfall variability (floods & droughts) | | • **Historical map of Rhenostervlei Farm dated AD 1815, illustrates a wagon road, suggesting the use of donkeys and wagons (Land Surveyor General 1964).** | • Severe drought conditions during the middle AD 1890s (Vogel 1989).<br>• Precipitation maxima ca.90 cal BP (ca. AD 1860) (Stager et al. 2012). |
| | **RV3 core basal date at 108 cm deep ca. AD 1795 (modelled calibrated)** | | | |
| | Cold and wet Little Ice Age (LIA) (AD 1400-1800) LIA maximum cooling (AD 1700) (Tyson et al. 2000) | Increase in Asteraceous shrubs (incl. Renosterbos) and grasses. Exotics introduced (grains, pines, wine grapes, waterblommetjies) | • Stock farming and wheat cultivation in Swartland in 18th C (River Health Programme, 2004).<br>• **A structural artefact/historical ruin (like a floor oven for making a fire) on the cliff ("krans") near the river possibly from the 17th/18th century suspected nomadic people (Gildenhuys pers. comm. 2019).** | • Precipitation maxima ca.330 cal BP (ca. AD 1620) (Stager et al. 2012). |

## 6.2 Vegetation survey, modern pollen estimation and rainfall history of Rhenostervlei Farm site

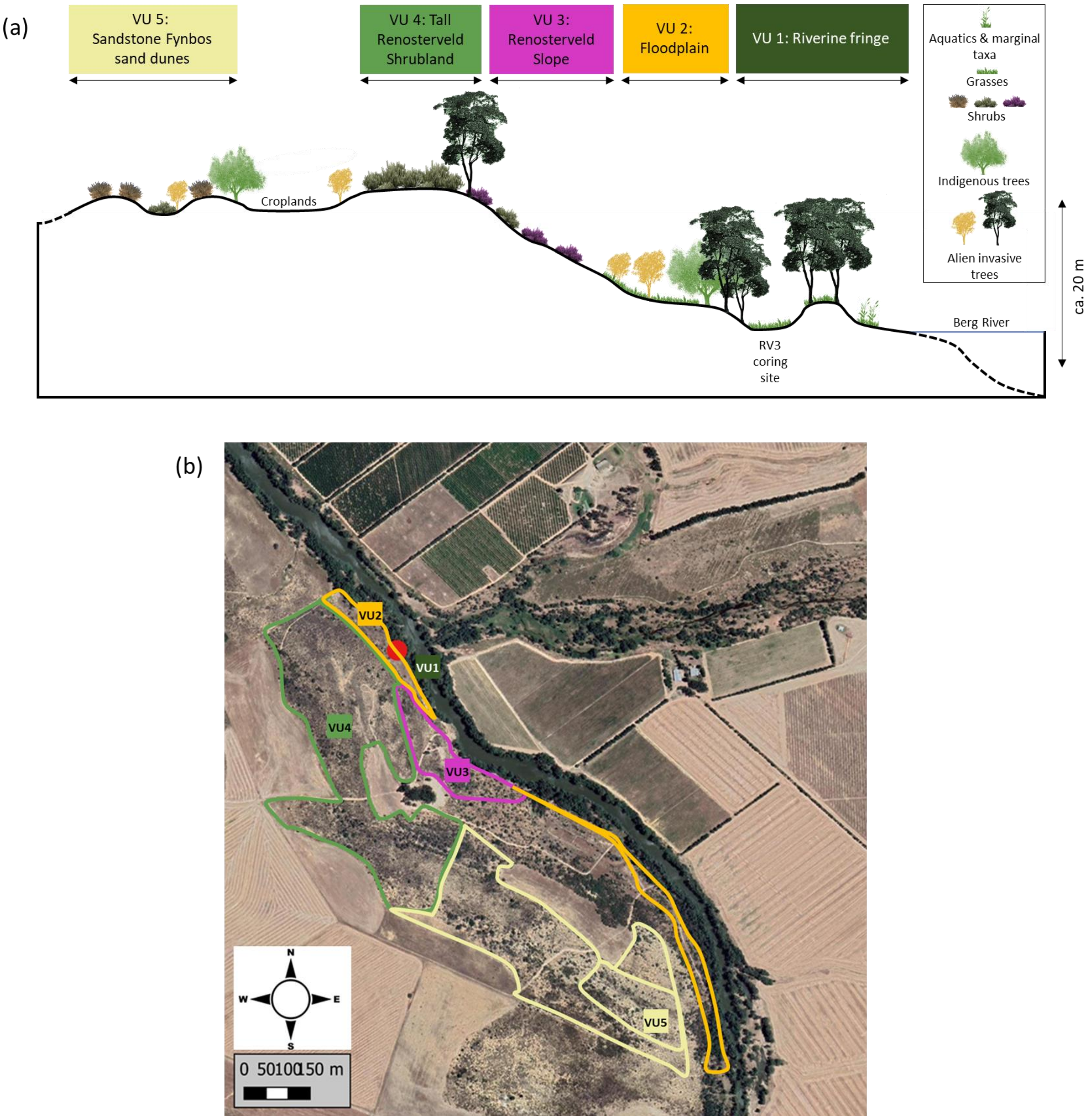
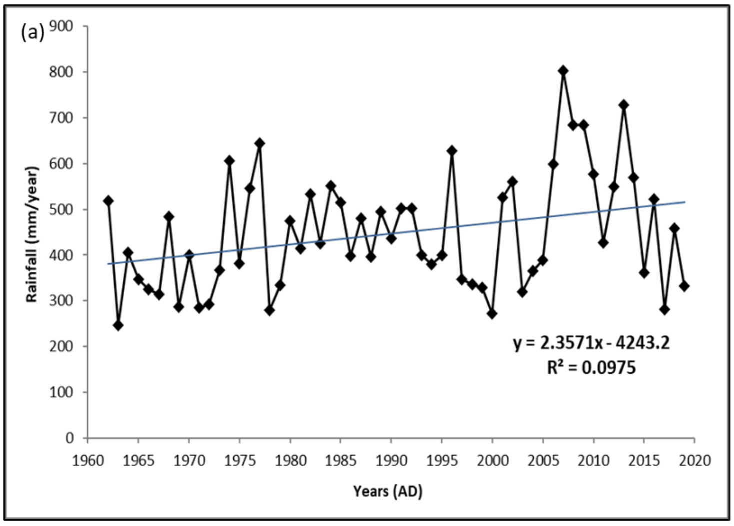



*Figure 6.1: (a) Diagrammatic representation of the profile of vegetation units (VU 1-5) that were surveyed. (b) Google earth map of selected area at Rhenostervlei Farm in 2020. Demarcated areas represent vegetation units that were chosen according to the most dominant vegetation type.*

Given the wetland basin diameter (±16-28 m diameter), it was assumed that most of the pollen in the RV3 core would have originated locally (Table 3.1) and therefore the site-based vegetation survey would provide information on the modern vegetation at the sites which would be compared with the pollen assemblage in the surface samples. Based on previous work in patchy landscapes it was estimated that most pollen would originate from within 300 m of the site, with a smaller component of regional pollen originating further away (Jacobson Jr and Bradshaw, 1981; Sugita, 1994). However, since the RV3 site is an area of low-lying ground adjacent to the Berg River, which is subject to flooding and therefore includes river sediments, the pollen record is also likely to represent a regional pollen signal and distal water borne components (e.g. geochemistry, MS) and/or be subject to periodic scouring by flood water. Moreover, RV3 wetland also experiences seasonal waterlogging even when the river does not overflow and as it surrounded by productive agricultural lands (Figure 6.1) it is subject to nutrient-loading and algal blooms during wet phases. The five vegetation units (VU1-VU5) in the vicinity of the RV3 wetland at Rhenostervlei Farm are shown in Figure 6.1, summarised in Table 6.2 and listed in detail in Appendix Table 10.5, which shows the most abundant taxa in each vegetation unit determined during the vegetation survey. Many of the 55 species that were identified during the modern vegetation survey are from families that are indistinguishable in the pollen record below family level (e.g. Proteaceae, Restionaceae, Poaceae, Cyperaceae, Asteraceae). Therefore, only 29 families and one genus (Asteraceae *Stoebe/Elytropappus*-type) could be identified in the pollen subsampled from the surface sample.

***Table 6.2: Vegetation units at RV3 wetland, Rhenostervlei Farm study site***

| Vegetation Unit | Description |
|---|---|
| VU 1: Riverine fringe | Riverine gallery forest (corridor) with IAP, blue gum, *Eucalyptus camaldulensis,* (Myrtaceae), as the dominant species and co-dominant Poaceae (*Ehrharta longiflora, Lolium perenne, Cynodon dactylon*). There were notable Thicket taxa (*Searsia, Olea europaea subssp. africana, Kiggelaria africana*). This is where the RV3 wetland is located. |
| VU 2: Floodplain | Floodplain with Poaceae (*Ehrharta longiflora*) and Cyperaceae (*Isolepis* spp.) as co-dominant and scattered IAP, wattle, *Acacia saligna* (Fabaceae) in the transition zone. |
| VU 3: Renosterveld Slope | A transitional matrix of sparse, short renosterveld with *Drosanthemum asperulum* (Aizoaceae) and a notable amount of bare ground on rocky slopes, south-east of the wetland. There is an erosion gully on the margins of the ecotone. The bottom of the cliff is hollowed out (like a cave) and a structure on the ground looks like an oven, ideal for making a fire and cooking. |
| VU 4: Tall Renosterveld Shrubland | South and south-west of the wetland is an area of 1 m tall shrubland dominated by *Passerina* (Thymelaeaceae) and *Spergula arvensis* (Caryophyllaceae) and unpalatable Renosterbos (*Elytropappus rhinocerotis*) (Asteraceae), which has not been burnt in the last 15 years. There is encroachment of *Acacia saligna* nearby towards the south of the BB plots. |
| VU 5: Sandstone Fynbos sand dunes | South-east of the Renosterveld vegetation is Fynbos Shrubland on fixed dunes: Top (crest) of the dunes were dominated by Proteaceae (mostly *Leucadendron* spp and *Leucospermum rodolentum*) and *Diospyros* spp with the bottom (trough) dominated by *Passerina*. Restios were present in both the troughs and crests. The area has burnt within the last 10 years. Remnant thicket (*Olea europaea subssp. africana*) with encroaching *Acacia saligna* on the margins. |



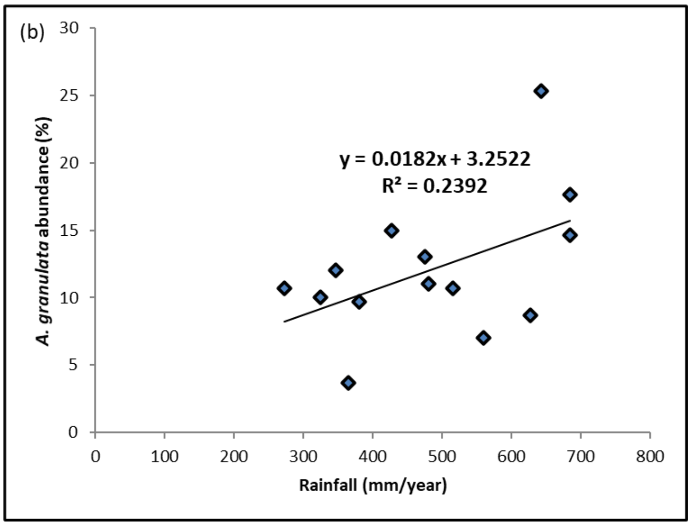




***Figure 6.2: (a) Rainfall data (mm/year) collected from Rhenostervlei Farm between AD 1962-2019. (b) Correlation ($R^2$=0.2392) between rainfall data and the abundance (%) of the (tyco)planktonic, freshwater diatom taxon, Aulacoseira granulata.***

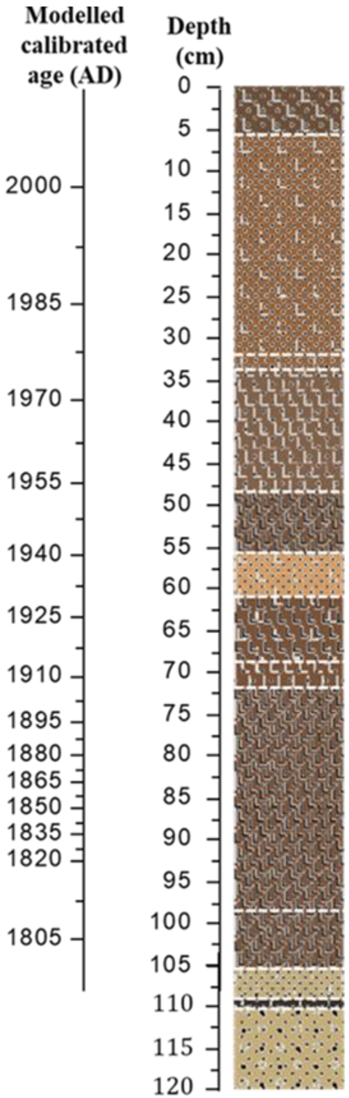
Figure 6.2a shows the variability in rainfall data collected at Rhenostervlei Farm between AD 1962-2019, with a mean annual rainfall of 450 mm/year. The general trend is increasing slightly over the 57-year period, with the highest annual rainfall (803 mm/year) received in AD 2007. The lowest recorded amount was 281 mm/year in AD 2017. Since the RV3 wetland is located next to the Berg River (Figure 6.1), it receives inputs when the river floods and it is a natural drainage area. It is therefore assumed that water turbidity is high with a high depositional rate at the site (108 cm in ca. 215 yrs). Based on the interpretation of diatom ecology (Table 4.2), as well as observations from the field during the vegetation survey and semi-structured interviews with the landowner (Gildenhuys pers. comm., 2021), the wetland fluctuates between 0 cm (drying out component) to being inundated and stagnant during wetter phases when the river breaches the embankment (ca. 10-20 cm water depth with a wet subaerial habitat) and floodplain (ca. 150 cm water depth). Furthermore, there was a noteworthy ($R^2$=0.2392) correlation between rainfall and the freshwater indicator diatom taxon, *Aulacoseira granulata* (Figure 6.2b). However, it should be noted that an absence of *Aulacoseira granulata* in the diatom record means it is not inundated and a range of epipelic and epilithic diatoms may occur (Gell et al., 2007; Kirsten, 2014; Stager et al., 2013, 2012; Taylor et al., 2007).

### 6.3 Sediment description and Loss on Ignition (LoI)

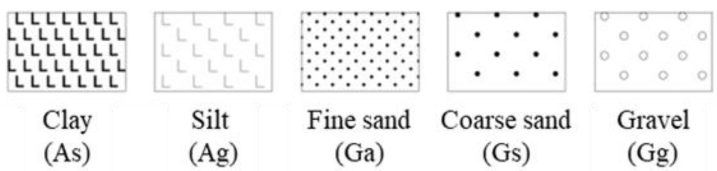
Results for sediment composition (Troels-Smith, 1955) and Loss on Ignition ($LoI_{550}$) (Bengtsson and Enell, 1986; Heiri et al., 2001; Walter E. Dean, 1974) of the RV3 core can be found in Table 6.3 and Figure 6.11, respectively. The RV3 core consisted of different shades of brown, with some areas of mottling and were described according to 13 sections. Details can be found in Table 6.3 with several different observations, which may relate to geochemical changes and the interpretation of ecosystem services changes in the RV3 core. The lowest section of the core (110-120 cm) contained mostly fine sand, some coarse sand and gravel. The upper contact was sharp, and therefore, not suitable for micro-fossil analysis. A thin section (109-110 cm) above consisted of silt and fine sand with ligneous material and herbaceous roots, and a noticeably high concentration of macro-charcoal remains.

Water content of the sediments decreased from deeper to shallower depths, with the highest percentage water content at 92 cm (20.31%) and the lowest at 8 cm (0.65%) (Figure 6.11). The lower percentage of water found later in the core is due to the high proportion of fine sand found in these layers (0-33.5 cm), which is more porous due to larger particle size compared to the deeper sediments consisting of more clay and silt. The percentage water content generally followed the trend for percentage organic carbon ($LoI_{550}$), except in the top 0-10 cm. Generally, the sediments contained low amounts of organic carbon ($LoI_{550}$ mean 4.1%) (Figure 6.11), with the lowest amount (1.3% at 58 cm) at a depth where the sediment consisted of predominantly fine sand, some silt and ca. 10% coarse sand (Table 6.3). When organic carbon was its highest (7.6% at 104 cm) the sediment consisted of predominantly clay with ligneous material and herbaceous roots. Inorganic carbon ($LoI_{1000}$, in the form of carbonate) was also generally low (mean 0.7%) and ranged from 0.07 (30 cm) to 1.2% (50 cm). There was a general trend of decreasing organic carbon up the core, while inorganic carbon was more variable over time (Figure 6.11).

***Table 6.3: Troels-Smith stratigraphy and Munsell soil colour description of sediments in the RV3 core. White dashed line shows the gradational upper contact.***




| Section depth (cm) | Troels-Smith Description | Munsell soil colour and code | Notes |
|---|---|---|---|
| 0-6 | Ga 2; Ag 2; Tl+3 | Olive brown (2.5Y, 4/4) | Pieces of macro-charcoal and herbaceous or ligneous pieces. |
| 6-32 | Ga 3; Ag 1; Tl+2 | Brown (10YR, 4/6) mottled with light olive brown (2.5Y,5/6) | Charred lignified piece (>1 cm x 2 mm). |
| 32-33.5 | Ga 3; Ag 1 | Light olive brown (2.5Y, 5/6) | |
| 33.5-48.5 | Ag 3; Ga 1; Tl+1 | Brown (10YR, 4/3) mottled with light olive brown (2.5Y, 5/6) and dark yellowish brown (10YR, 4/4) | Roots and charred lignified pieces (1 cm x 1 mm). |
| 48.5-56 | As 4; Tl+2 | Olive brown (2.5Y, 4/4) mottled with dark yellowish brown (10YR, 4/4) | Oxidation layers (reddish bands at 52-53 cm). |
| 56-61 | Ga 3; Ag 1 | Light yellowish brown (2.5Y, 6/4) | Evidence of <10% coarse sand (Gs). |
| 61-69 | As 3; Ag1; Th+1 | Olive brown (2.5Y, 4/4) | |
| 69-72 | Ag 3; Ga 1 | Olive brown (2.5Y, 4/4) mottled with 75% dark yellowish brown (10YR, 4/4) | |
| 72-98 | As 4; Tl+3 | Olive brown (2.5Y, 4/3) mottled with dark yellowish brown (10YR, 4/4) | High concentration of macro-charcoal; ca. 10% silt (Ag). |
| 98-105.5 | As 4; Tl+3; Th+1 | Dark olive brown (2.5Y, 3/3 - base colour) mottled with dark yellowish brown (10YR, 4/4) | |
| 105.5-109 | Ga 3; Ag 1; Tl+4 | Light yellowish brown (2.5Y, 6/4) | Lignified pieces (2-3 mm) not as charred. |
| 109-110 | Ga 2; Ag 2; Tl+3; Th+1 | Dark olive brown (2.5Y, 3/3) | High concentration of macro-charcoal. |
| 110-120 | Ga 2; Gg 1; Gs 1; Tl+2 | Light yellowish brown (2.5Y, 6/4) | Gravel pieces and several lignified pieces. |



## 6.4 Chronology of RV3 - Constant Rate of Supply (CRS) model for $^{210}$Pb dating and Bacon Age-Depth model

AMS radiocarbon dating at the basal depth 108 cm resulted in an age of 118 ± 30 yr BP. The probability distribution is multi-modal (Appendix Figure 10.13) and the modelled calibrated basal age was ca. AD 1795. This age is consistent with the pollen spectrum of the RV3 core, which shows *Eucalyptus, Pinus, Acacia* and *Cerealia* pollen present in lower depths. Historical records report that alien invasive plants and crops were introduced in the region in the 17$^{th}$ Century (Table 5.1 and Table 6.1). The trend in sediment accumulation rate over time was consistent between all three statistical modelling methods and showed a general increasing trend up the RV3 core. See results of the Constant Rate of Supply (CRS) model (Appendix Table 10.6; Figure 6.3); Slope Regression model in Appendix 10.6.3; and the Bacon Age-Depth model (Figure 6.4 and Appendix Figure 10.13).

Measurable quantities of $^{210}$Pb were found in the RV3 core between 0-90 cm, below which no measurable $^{210}$Pb was found suggesting that sediments below 90 cm were older than 150 years (Table 10.6a). The lowest observed activity (1.12 DPM/g) of the RV3 core at 95-100 cm (sediment section no. 33) was assumed as the background $^{210}$Pb level. The CRS model was applied to determine the mass accumulation rate since the RV3 core did not experience constant sediment accumulation over time (Table 10.6; Figure 6.3). Average sediment accumulation rates from the CRS model were calculated according to the pollen zones (explained in section 6.5.1): a slower sedimentation rate (0.2 g/cm$^2$/year) from 45-90 cm, followed by a higher rate (0.24 g/cm$^2$/year) from 9-45 cm; and an ever higher sediment accumulation rate (0.26 g/cm$^2$/year) from 0-9 cm.  Furthermore, using Bacon in R, an age-depth model was produced with the AMS radiocarbon and $^{210}$Pb dates (Figure 6.4). The sedimentation appeared to show no reversals and the accumulation rates are reasonable estimates for the part of the RV3 core that was $^{210}$Pb dated. Average sediment accumulation rates showed a similar increasing trend over time: a consistent sediment accumulation (0.62 cm/year) from 90-108 cm; a slower sedimentation rate (0.37 cm/year) from 45-90 cm; followed by a higher sediment accumulation rate (0.81 cm/year) from 9-45 cm; and an even higher sediment accumulation rate (1.14 cm/year) from 0-9 cm.

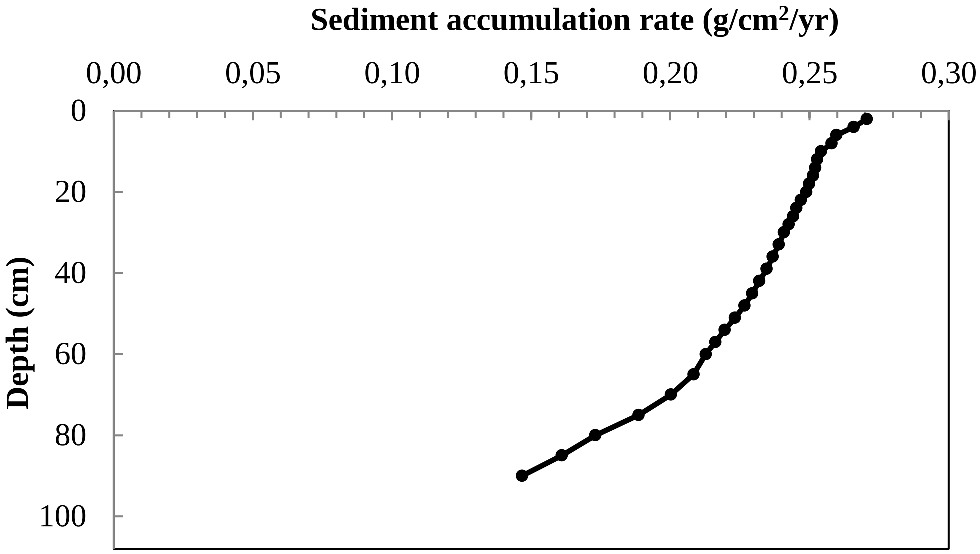



*Figure 6.3: Constant Rate of Supply (CRS) model showing sediment accumulation rate vs. depth at the bottom of each sediment section.*

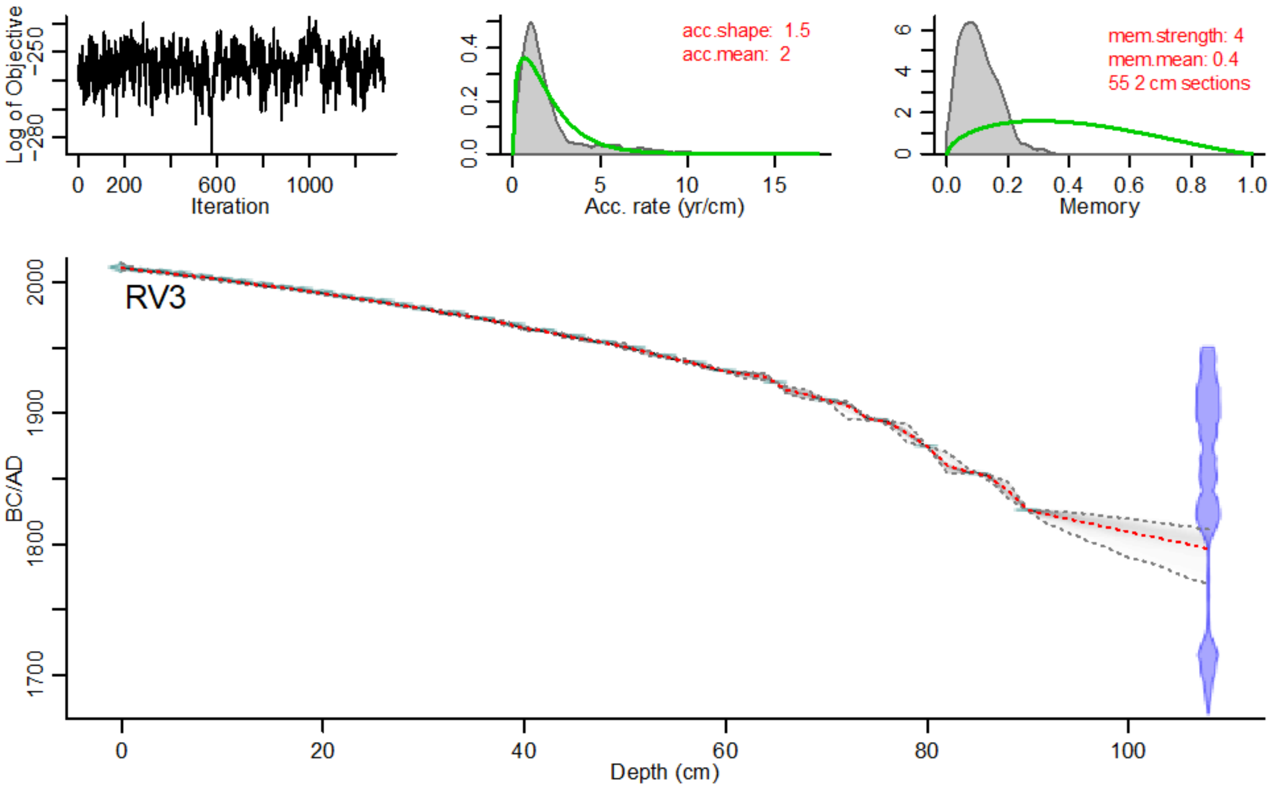



*Figure 6.4: Age-depth model developed using Bacon for samples from 0 to 108 cm for the RV3 core including the following: the date that the core was extracted (AD 2011), one calibrated AMS radiocarbon date (ca. AD 1879-1925) and thirty-one $^{210}$Pb dates with SD error bars.*

## 6.5 Palaeoenvironmental proxy results for RV3 core

### 6.5.1 Pollen results for RV3: A proxy for plant biodiversity

Within the pollen spectrum over the ca. 215-year timescale, on average, the most abundant pollen was that of Renosterbos (mean 28% and mean 11 751 grains/cm$^3$). This is a dominant unpalatable plant species common in Renosterveld vegetation. Equally abundant in the pollen spectrum was pollen grouped into Renosterveld and Fynbos (mean 28% and mean 10 326 grains/cm$^3$). The next abundant was Thicket (mean 16% and 6 516 grains/cm$^3$), then Exotics (mean 13% and 4 362 grains/cm$^3$), Fynbos (mean 9% and 3 423 grains/cm$^3$) and Grassland (mean 4% and 1 339 grains/cm$^3$). Changes in the dominance of these vegetation types occurred over time (Figure 6.5 and Figure 6.6) and are described below. Unknown pollen grains were relatively low (mean 2.5% and 862 grains/cm$^3$) in abundance as most pollen grains were identifiable in the RV3 record.

#### *6.5.1.1 Pollen zonation*

A zonation (cluster analysis) showed two statistically significant zones based on the square root transformed pollen data and the Bray-Curtis similarity data (Figure 6.6 and Table 5.3). Table 6.4 summarises the pollen zonation results, which depicts the distribution of both percentage pollen data (Figure 10.15a and Figure 10.16 a) and concentration pollen data (Figure 10.15b and Figure 10.16b) based on modelled calibrated ages and depth of the RV3 core. The zonation results for percentage and concentration pollen data were consistent since significantly different zones were determined at the same depths. The broken stick models of the pollen percentage and concentration data consistently indicated two significant zones because the value of the black line was above the critical value represented by the red line (Figure 10.15a and b). Zone R1 includes depths 108-45 cm and according to the RV3 age-depth model (Figure 6.4) it was dated as ca. AD 1795-1960. Zone R2 includes depths 45-0 cm dated as ca. AD 1960-2011. Additional sub-zoning was plotted within the significantly distinct zones (see dashed lines in Table 6.4 and Figure 6.5 and Figure 6.6) based on the cluster analysis. Although the sub-zones were not statistically significant, they were useful in order to describe other notable changes in the pollen stratigraphy over time. Therefore, Zone R1 was statistically different from Zone R2, and Sub-zones R1a (108-90 cm; ca. AD 1795-1830) and R1b (90-45 cm; ca. AD 1825-1960) were not significantly different from each other, nor were Sub-zone R2a (45-9 cm; ca. AD 1960-2005) and R2b (9-0 cm; ca. AD 2005-2011) (Figure 10.15).

***Table 6.4: Details of zonation for the RV3 pollen record. The solid line separating zones R1 and R2 indicates statistically significant zonation, while the dashed lines indicate sub-zones.***

| Pollen Sub-zone | Number of sub-sample depths analysed | Depth ranges of samples (cm) | Dates of pollen zone boundaries (AD) | Number of years |
|---|---|---|---|---|
| R2b | 4 | 0-9 | 2005-2011 | 6 |
| R2a | 12 | 9-45 | 1960-2005 | 45 |
| R1b | 16 | 45-90 | 1825-1960 | 135 |
| R1a | 6 | 90-108 | 1795-1825 | 30 |

#### *6.5.1.2 Pollen Sub-zone R1a (6 sub-sample depths; 108-90 cm; ca. AD 1795-1825)*

Sub-zone R1a, representing depths 108-90 cm (ca. AD 1795-1825), is characterized by pollen types that represent shrubland with Thicket elements (Figure 6.5 and Figure 6.6). The aggregate of pollen representing indigenous taxa (i.e. total sum of all terrestrial pollen taxa excluding exotics) had its highest mean of 95% in R1a, whereas exotics had its lowest mean of 5%. This result is also expressed as a mean ratio of indigenous:exotic pollen types, which showed the second highest average (mean ratio 20) in Sub-zone R1a. Asteraceae *Stoebe/Elytropappus*-type pollen (Renosterbos characteristic of Renosterveld vegetation) was the most abundant type in R1a (mean 33% and 20 478 grains/cm$^3$). It increased in both the percentage and concentration data and fluctuated slightly at a substantive abundance within the sub-zone. Generally, the total Renosterveld and Fynbos taxa increased steadily throughout Sub-zone R1a. Asteraceae long-spine type-1 was the most abundant taxa characteristic of Renosterveld and Fynbos vegetation and it was the third most abundant pollen type in Sub-zone R1a (mean 22% and 13 907 grains/cm$^3$).

Taxa characteristic of Fynbos vegetation were generally low (mean 8.5% and 4 850 grains/cm$^3$) compared to the other vegetation groups. Total Fynbos taxa increased over time. The most abundant Fynbos taxa were *Cliffortia* (Rosaceae) (mean 2.5% and 1 452 grains/cm$^3$), *Anthospermum* (Rubiaceae) (mean 2.1% and 1 041 grains/cm$^3$) and Ericaceae (mean 1.3% and 731 grains/cm$^3$). All three taxa showed an increase through the sub-zone with a noticeable decrease at the R1a/R1b sub-zone transition. Of the less abundant Fynbos taxa, Restionaceae, Proteaceae, *Agathosma* (Rutaceae) and *Diosma* (Rutaceae) were also present, but Restionaceae pollen was the only taxon that persisted whereas the others were absent at various times during R1a. *Diosma* pollen relative abundance was highest in Sub-zone R1a for the concentration data (967 grains/cm$^3$) in ca. AD 1820 (92 cm), which was just before the sub-zone R1a/R1b transition. Whereas *Agathosma* pollen was absent when the sub-zone R1a/R1b transition occurred (between ca. AD 1820 -1840 (92-88 cm)).

Taxa characteristic of Thicket vegetation included Oleaceae type-2 pollen (likely *Olea europaea subssp. Africana*) which was the second most abundant taxon in Sub-zone R1a. During ca. AD 1795-1830 (108-90 cm) it reached its highest mean value of 25% and 15 373 grains/$cm^3$. Similar to the abundant Asteraceae *Stoebe/Elytropappus*-type and Asteraceae long-spine type-1 it also increased throughout Sub-zone R1a. For the concentration data only, the greatest abundance of Oleaceae type-2 was just before the R1a/R1b transition (28 676 grains/$cm^3$ in ca. AD 1820 (92 cm)). Of the less abundant Thicket taxa, Oleaceae type-1, *Searsia, Maytenus oleoides* (Celastraceae) and *Dodonaea* (Sapindaceae) were present in the early part of the sub-zone. The following thicket taxa reached their highest relative abundance over the ca. 215-year palaeo-record for both percentage and concentration data: Oleaceae type-1 (4% in ca. AD 1795 (108 cm) and 2 140 grains/cm3 in ca. AD 1805 (102 cm)), *Searsia* (2% and 1 634 grains/cm3 in ca. AD 1815; 96 cm) and *Dodonaea* (2% and 2 140 grains/cm3 in ca. AD 1810; 100 cm). When compared to other vegetation groups, Poaceae pollen representative of Grassland-type patch was relatively low (mean 2% and 1 309 grains/$cm^3$) and showed a slight increase and a later decrease in abundance during the sub-zone. Poaceae pollen abundance was the lowest in R1a for the percentage data only (0.7%) in ca. AD 1820 (92 cm), which was just before the sub-zone R1a/R1b transition. Total exotic taxa were sparse in both the percentage (mean 5.1%) and concentration, (mean 3 196 grains/$cm^3$) data (Figure 6.5 and Figure 6.6).

Total Exotics increased slightly and then decreased again in the later parts of the sub-zone. The most abundant of the exotic taxon was *Pinus* (mean 4% and 2 446 grains/$cm^3$). Less abundant exotics, *Eucalyptus* and *Acacia*, were sometimes present in the sub-zone, but *Cerealia* (cereal crop pollen - 2% and 545 grains/$cm^3$) was only present once in ca. AD 1800 (104 cm).

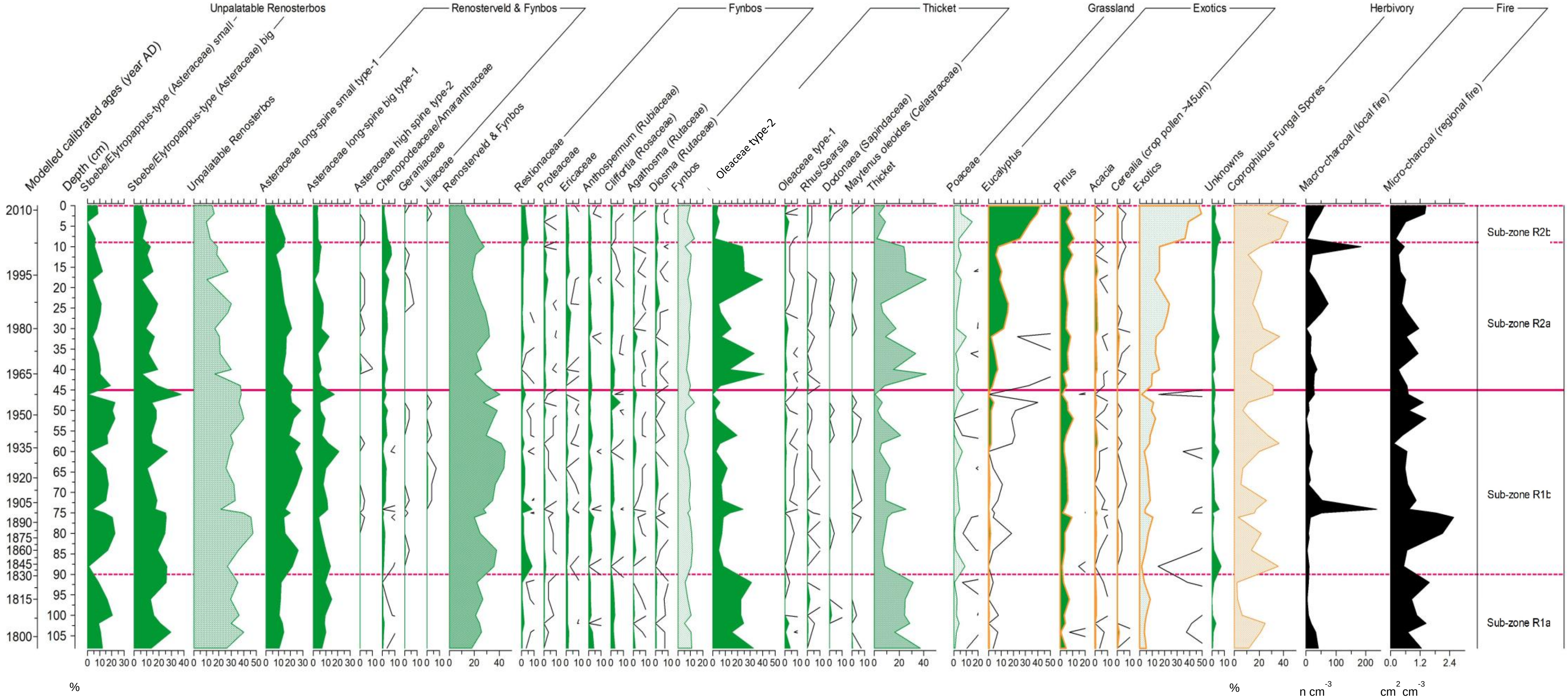
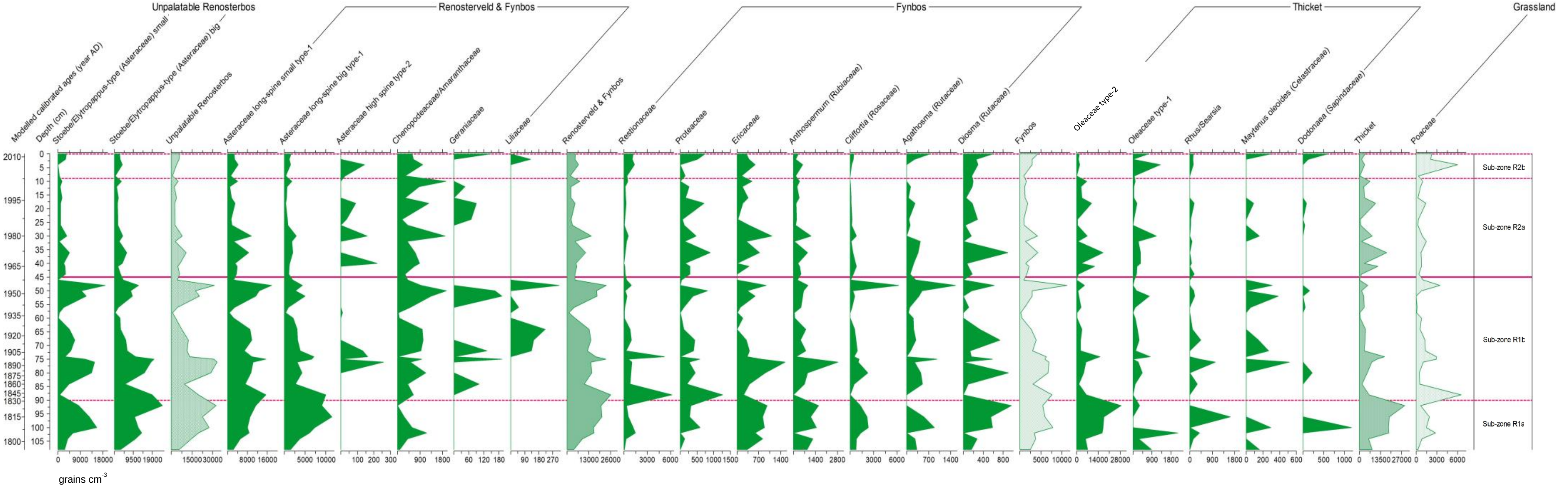



*Figure 6.5: Percentage pollen diagram with selected palaeo-proxies, (coprophilous fungal spores, and macro- and micro-charcoal) from the RV3 core at Rhenostervlei Farm, Middle Berg River Catchment, South Africa. The pollen sum included terrestrial and unknown pollen. x10 exaggeration is shown for low taxa. Summary graphs with pollen taxa grouped into ecological groupings of major vegetation types. Zonation is calculated by cluster analysis based on 70% minimum resemblance levels – statistically significant zones are indicated by the solid line (Zone R1 and Zone R2) whereas dashed lines show sub-zones (R1a, R1b, R2a, R2b) to the far right of the diagram. Modelled calibrated ages that were determined using an age-depth model (Bacon) are shown on the left.*



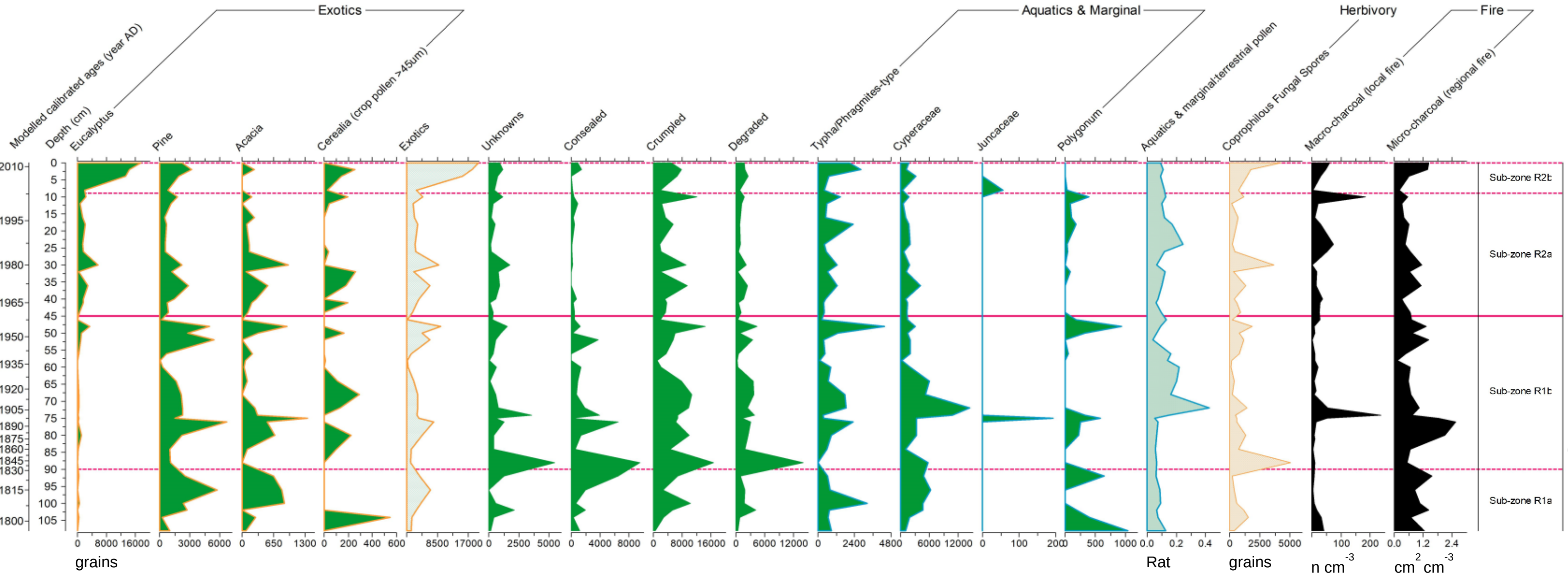




*Figure 6.6: Concentration pollen diagram with coprophilous fungal spores, and macro- and micro-charcoal from the RV3 core. Pollen taxa are grouped into major vegetation types. Statistically significant zones are indicated by the solid line whereas dashed lines show sub-zones (R1a, R1b, R2a, R2b). Modelled calibrated ages are shown on the left.*

On average, the 'aquatics and marginal:terrestrial pollen' ratio was the lowest (mean ratio 0.09%) over the ca. 215-year RV3 palaeo-record during Sub-zone R1a (Figure 6.6). It showed a slight decline but was mostly stable in terms of its abundance throughout the zone. Cyperaceae was the most abundant taxon (mean 3900 grains/$cm^3$) characteristic of aquatics and marginal pollen types as it increased throughout the sub-zone. *Typha*/*Phragmites*-type and *Polygonum* were less abundant and experienced great variability. *Polygonum* reached its highest relative abundance (1 043 grains/$cm^3$) in ca. AD 1795 (108 cm). Concealed, crumpled and degraded pollen grains were excluded from the terrestrial pollen sum since they were unidentifiable. However, Figure 6.6 shows that the concentration patterns for the concealed (mean 2 046 grains/$cm^3$), crumpled (mean 4 736 grains/$cm^3$) and degraded (mean 1 656 grains/$cm^3$) pollen grains were variable and increased towards the sub-zone R1a/R1b transition.

#### *6.5.1.3 Pollen Sub-zone R1b (16 sub-sample depths; 90-45 cm; ca. AD 1825-1960)*

Sub-zone R1b, representing depths 90-45 cm (ca. AD 1825-1960), like the previous sub-zone is characterized by pollen taxa that represent shrubland and Thicket vegetation (Figure 6.5 and Figure 6.6). The aggregate abundance of indigenous pollen taxa reached its second highest mean of 93% in R1b, and that of exotics reached its second lowest mean of 7%. Despite the increase in exotics compared to the previous sub-zone, the indigenous:exotics taxa ratio was the highest (ratio 68) in ca. AD 1843 (88 cm) and the second highest (ratio 64) occurred just before the R1b/R2a zones transition at ca. AD 1960 (46 cm).

Asteraceae *Stoebe/Elytropappus*-type pollen characteristic of Renosterveld vegetation was the most abundant type in R1b (mean 34% and 15 554 grains/$cm^3$). In the percentage data it fluctuated throughout the sub-zone but in the concentration data, fluctuations in abundance were more pronounced with the lowest concentration (853 grains/$cm^3$) in ca. AD 1940 (58 cm). Asteraceae *Stoebe/Elytropappus*-type was its highest in Sub-zone R1b for both the percentage and concentration data: 47% in ca. AD 1880 (80 cm) and 32 993 grains/$cm^3$ in ca. AD 1895 (76 cm). Total Renosterveld and Fynbos taxa varied at a relatively high abundance throughout Sub-zone R1b, with a few unprecedented peaks and troughs more evident in the concentration pollen data. Again, Asteraceae long-spine type-1 was the most abundant pollen type characteristic of Renosterveld and Fynbos vegetation and it was the second most abundant pollen type on average (mean 32% and 12 808 grains/$cm^3$). Poaceae pollen, representative of Grassland-type patch in the patch-mosaic landscape, remained low during R1b (mean 3% and 1 473 grains/$cm^3$). In the concentration data only, its abundance was the highest (6 490 grains/$cm^3$) just after the R1a/R1b transition in ca. AD 1840 (88 cm) and lowest (92 grains/cm$^3$) in ca. AD 1940 (58 cm). It was completely absent in R1b in ca. AD 1950 (52 cm). Renosterveld and Fynbos taxa which were even less abundant such as Asteraceae long-spine type-2 (mean 0.1% and 35 grains/cm$^3$), Geraniaceae (mean 0.11% and 49 grains/cm$^3$) and Liliaceae pollen (mean 0.14% and 54 grains/cm$^3$), were present in the RV3 core for the first time during Sub-zone R1b.

On average, taxa characteristic of Fynbos vegetation remained relatively low (mean 9% and 4 166 grains/cm$^3$), with a noticeable decline during the middle period of Sub-zone R1b followed by a dramatic incline evident in the concentration data. This noticeable peak in Fynbos pollen taxa (13% and 11 225 grains/cm$^3$) occurred in ca. AD 1955 (48 cm), with increases in *Cliffortia* and *Agathosma* contributing significantly to this trend. Interestingly, Restionaceae was its highest in R1b for both the percentage and concentration data just after the R1a/R1b transition: 9% and 6 181 grains/cm$^3$ in ca. AD 1840 (88 cm). At this particular period other Fynbos taxa were completely absent (i.e. *Cliffortia*, *Anthospermum*, *Agathosma*) but Proteaceae and Ericaceae pollen were still present.

Despite remaining the most abundant taxon characteristic of Thicket vegetation, Oleaceae type-2 pollen exhibited a dramatic decline during Sub-zone R1b, evident in both the percentage and concentration data: mean 8% and 3 954 grains/cm$^3$, respectively. It fluctuated noticeably, experiencing highs and lows during ca. AD 1830-1960 (90-45 cm). Less abundant Thicket taxa, such as Oleaceae type-1 and *Searsia,* remained present throughout the sub-zone whereas *Maytenus oleoides* and *Dodonaea* were only present a few times.

On average, pollen characteristic of Exotic taxa (*Eucalyptus*, *Acacia*, *Pinus and Cerealia*) remained sparse (mean 6.8% and 3 121 grains/cm$^3$) during Sub-zone R1b. Although there was a noticeable increase in *Eucalyptus* pollen (mean 1.1% and 537 grains/cm$^3$) during this time, *Pinus* (mean 5% and 2 213 grains/cm$^3$) was the most dominant of the exotic taxa in this sub-zone.

The 'aquatics and marginal:terrestrial pollen' ratio reached its highest ratio of 0.4 in ca. AD 1905 (72 cm) and lowest ratio of 0.04% in ca. AD 1950 (52 cm) (Figure 6.6). It had the highest average (mean ratio 0.13) compared to the other sub-zones over the ca. 215-year RV3 palaeo-record. Cyperaceae remained the most abundant taxon (mean 4 071 grains/cm$^3$) characteristic of aquatics and marginal vegetation as it increased throughout the sub-zone. Interestingly, *Phragmites*-type was absent in the pollen record in ca. AD 1840 (88 cm) which is just after the Sub-zone R1a/R1b transition. *Polygonum* was also absent around the same time in ca. AD 1840-1860 (88-84 cm). Compared to the other taxa, Juncaceae pollen was low in abundance (mean for the RV3 core was 7 grains/cm$^3$) and occurred for the first time in the RV3 palaeo-record in R1b in ca. AD 1900 (75 cm) (193 grains/cm$^3$). *Phragmites*-type reached its highest relative abundance (4 365 grains/cm$^3$) and *Polygonum* reached its second highest abundance (935 grains/cm$^3$) in ca. AD 1955 (48 cm).

On average, the abundance patterns decreased for the concealed grains (mean 2 005 grains/cm$^3$) but increased substantially for the crumpled (mean 7 079 grains/cm$^3$) and degraded pollen (mean 2 907 grains/cm$^3$) during R1b (Figure 6.6). The pattern was highly variable throughout the ca. 215 years, with the highest relative abundance of concealed (9 503 grains/cm$^3$), crumpled (16 682 grains/cm$^3$) and degraded pollen (13 973 grains/cm$^3$) occurring in ca. AD 1840 (88 cm), just after the sub-zone R1a/R1b transition. Degraded and concealed pollen reached their lowest abundance at ca. AD 1940 (58 cm). A noteworthy peak in crumpled pollen (14 343 grains/cm$^3$) occurred at ca. 1955 (48 cm) just before the zonal transition of R1b to R2a, which was a significant transition in the pollen assemblage in the early ca. AD 1960s.

#### *6.5.1.4 Pollen Sub-zone R2a (12 sub-sample depths; 45-9 cm; ca. AD 1960-2005; ca. 45 years)*

Sub-zone R2a represents depths of 45-9 cm (ca. AD 1960-2005). Like previous sub-zones, there is a relatively large abundance of pollen taxa characteristic of shrubland and Thicket vegetation (Figure 6.5 and Figure 6.6). Thus, the aggregate indigenous pollen taxa remained high (mean 84%) in R2a. However, exotics abundance doubled (mean 16%) since the previous sub-zone. The R2a indigenous:exotics pollen taxa experienced its second lowest mean ratio of 6.

Asteraceae *Stoebe/Elytropappus*-type pollen, which is characteristic of Renosterveld vegetation, declined significantly during R2a (mean 23% and 4 833 grains/cm$^3$). This trend was more pronounced in the concentration data compared to the percentage data and provides evidence of the Fagerlind effect. Asteraceae long-spine type-1, which is characteristic of Renosterveld and Fynbos vegetation, remained the most abundant pollen type (mean 21% and 4 877 grains/cm$^3$) in Sub-zone R2a. However, Asteraceae long-spine type-1 pollen exhibited a similar decline to the Renosterveld taxon (Asteraceae *Stoebe/Elytropappus*-type pollen). Chenopodiaceae/Amaranthaceae pollen remained less abundant and fluctuated as it increased during R2a (mean 3.7% and 820 grains/cm$^3$). Geraniaceae abundance remained low (mean 0.12% and 17 grains/cm$^3$) and Liliaceae pollen was completely absent in Sub-zone R2a (ca. AD 1960-2005).Taxa characteristic of Fynbos vegetation also remained relatively low (mean aggregate of 8.7% and 1 924 grains/cm$^3$) and continued to fluctuate in a relatively predictable manner. However, *Cliffortia* (mean 1.4% and 314 grains/cm$^3$) and Restionaceae (mean 1.1% and 222 grains/cm$^3$) pollen showed a noticeable decline in R2a. *Anthospermum* pollen was its lowest in ca. AD 1980 (32 cm) (0.3% and 43 grains/cm$^3$) whereas *Diosma* pollen abundance was highest in R2a for the percentage data (3%) in ca. AD 1990 (24 cm). Like *Diosma* pollen, Ericaceae pollen reached also its highest relative abundance in R2a for the percentage data only (4% in ca. AD 1985 (26 cm)), whereas for the concentration data, its highest relative abundance was in R1b (1 547 grains/cm$^3$ in ca. AD 1895 (76 cm)) providing evidence of the Fagerlind effect. Ericaceae pollen was completely absent in the RV3 core on four occasions: in ca. AD 1925 (64 cm) and ca. AD 1965 (40 cm), but most noticeably, just after and before the transition of zone R1b/R2a (ca. AD 1960 (46-44 cm). Poaceae pollen representing Grassland patches of vegetation remained low during R2a (mean 3.7% and 733 grains/cm$^3$).

Oleaceae type-2 pollen, which is characteristic of Thicket vegetation, increased substantially (mean 20% and 5 333 grains/cm$^3$) in Sub-zone R2a compared to the previous sub-zone. It was abundant in the early part of the sub-zone, declined in the middle and then increased again in the later part of the sub-zone. This pattern was consistent in both the percentage and concentration data. Less abundant Thicket taxa persisted. However, *Searsia, Maytenus oleoides* and *Dodonaea* were absent in the later part of the sub-zone (from ca. AD 1995 (16 cm) onwards).

On average, pollen characteristic of Exotic taxa (*Eucalyptus, Acacia, Pinus and Cerealia*) increased during Sub-zone R2a, mostly noticeable in the percentage pollen data (mean 15.5%) compared to the concentration pollen data (3 398 grains/cm$^3$). Trends in exotics showed an increased abundance during early parts of the sub-zone and declined in the later part. Unlike in Sub-zone R1b where *Pinus* was the most dominant exotic taxon, in Sub-zone R2a *Eucalyptus* (mean 8.4% and 1 846 grains/cm$^3$) was dominant. The 'aquatics and marginal:terrestrial pollen' ratio decreased slightly compared to the previous sub-zones and it was the second highest (mean ratio 0.12) over the ca. 215-year RV3 palaeo-record (Figure 6.6). The ratio increased slightly (ratio 0.25) towards the middle in ca. AD 1990 (24 cm).

On average, the abundance patterns for concealed (mean 324 grains/cm$^3$), crumpled (mean 4 811 grains/cm$^3$) and degraded (mean 1 173 grains/cm$^3$) pollen decreased, with the most dramatic decrease evident in the concealed grains. They fluctuated at a low level throughout Sub-zone R2a (Figure 6.6).

#### *6.5.1.5 Pollen Sub-zone R2b (4 sub-sample depths; 9-0 cm; ca. AD 2005-2011; ca. 8 years)*

Sub-zone R2b, representing depths 9-0 cm (ca. AD 2005-2011), showed a drastic decline in the aggregate of pollen representing local biodiversity (lowest mean of 57%) in R2b. Because exotics were at their highest mean of 43%, the ratio of indigenous:exotics was at its lowest (mean ratio 1) during this period. Asteraceae *Stoebe/Elytropappus*-type pollen, which is characteristic of Renosterveld vegetation, declined further during R2b (mean 13% and 4 204 grains/cm$^3$) (Figure 6.5 and Figure 6.6). The percentage trend decreased during this sub-zone whereas the trend in concentration increased over time, providing evidence of the Fagerlind effect. Asteraceae long-spine type-1, which is characteristic of Renosterveld and Fynbos vegetation, remained the most abundant taxa (mean 14% and 3 834 grains/cm$^3$) in Sub-zone R2b but generally declined in abundance relative to the previous sub-zone. Like Asteraceae *Stoebe/Elytropappus*-type pollen, the trend in the percentage of Asteraceae long-spine type-1 pollen decreased during this sub-zone whereas the trend in its concentration increased over time. Chenopodiaceae/Amaranthaceae, Asteraceae long-spine type-2 and Geraniaceae remained low in abundance and Liliaceae pollen appeared again in Sub-zone R2b in ca. AD 2010 (2 cm). Taxa characteristic of Fynbos vegetation increased slightly (mean aggregate of 9.8% and 2 803 grains/cm$^3$), with an initial decreasing trend and an increasing trend later during Sub-zone R2b. *Agathosma* was noticeably absent when the sub-zone R2a/R2b transition occurred between ca. AD 2000-2010 (10-4 cm). Restionaceae was generally higher than in the previous sub-zone and Ericaceae declined throughout Sub-zone R2b. Other Fynbos taxa (Proteaceae, *Anthospermum*, *Cliffortia*, *Agathosma* and *Diosma*) showed trends that increased in the later part of the sub-zone. Poaceae pollen characteristic of Grassland in the patch-mosaic landscape increased substantially during Sub-zone 2b (mean 7.7% and 2 669 grains/cm$^3$). A high peak in Poaceae pollen was evident in both the percentage (15%) and concentration data (6 017 grains/cm$^3$) during ca. AD 2010 (4 cm).

Oleaceae type-2 pollen (mean 4% and 1 256 grains/cm$^3$), which is characteristic of Thicket vegetation, decreased substantially compared to the previous sub-zone and fluctuated at a relatively low level throughout Sub-zone R2b (Figure 6.5 and Figure 6.6). Oleaceae type-2 pollen reached its lowest abundance in ca. AD 2005 (8 cm) for both the percentage (1%) and concentration data (111 grains/cm$^3$). Noticeably, *Searsia* pollen was absent at the sub-zone R2a/R2b transition between ca. AD 1995-2005 (16-8 cm)) (ca. 10 years). Similarly, *Maytenus oleoides* pollen was absent in ca. AD 1995-2010 (16-2 cm) (ca. 15 years around sub-zone R2a/R2b transition).

On average, Exotic taxa showed a dramatic increase during Sub-zone R2b, in both the percentage (mean 43.2%) and concentration of pollen data (13 970 grains/cm$^3$). Initially, *Acacia, Pinus and*

*Cerealia* increased and then decreased in the later part of the sub-zone, whereas *Eucalyptus* continued to increase throughout (Figure 6.5 and Figure 6.6). *Eucalyptus* pollen relative abundance was the highest in ca. AD 2011 (0 cm) (R2b) for both the percentage data (42%) and concentration data (17 561 grains/cm$^3$). The 'aquatics and marginal: terrestrial pollen' ratio decreased further to its second lowest mean of 0.1 in Sub-zone R2b and followed a stable trend throughout the sub-zone (Figure 6.6).

The abundance of concealed (mean 608 grains/cm$^3$), crumpled (mean 5 320 grains/cm$^3$) and degraded (mean 1 753 grains/cm$^3$) pollen increased. The trend showed an increase early in Sub-zone R2a which later decreased.

### 6.5.2 Macro-charcoal and micro-charcoal as proxies for local and regional fire

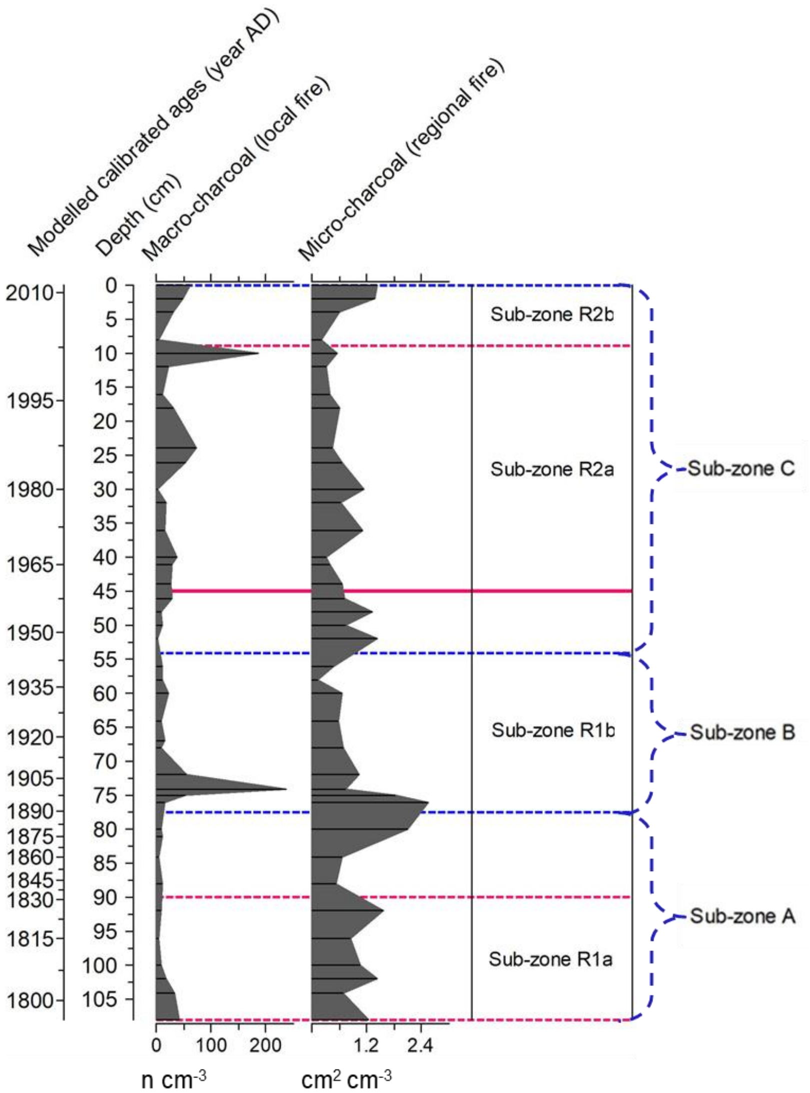



*Figure 6.7: Macro-charcoal (local fire) and micro-charcoal concentration (regional fire) for the RV3 core.*

In Sub-zone R1a (ca. AD 1795-1825; 108-90 cm), macro-charcoal was on average the lowest (mean 19 particles/$cm^3$) compared to subsequent sub-zones. It showed a high relative abundance in early parts of the sub-zone and declined halfway through and remained stable at a low level. However, the relative abundance of micro-charcoal was on average the highest (mean 1.1 $cm^2/cm^3$) in Sub-zone R1a and showed a fluctuating trend that increased throughout the sub-zone.

In Sub-zone R1b (ca. AD 1830-1960; 90-45 cm), macro-charcoal was on average the second lowest (mean 29 particles/$cm^3$) compared the other sub-zones. It showed a low stable abundance in the early parts of the sub-zone and increased dramatically to its highest relative abundance (236 particles/$cm^3$) over the ca. 215-year palaeo-record in ca. AD 1900 (74 cm). Macro-charcoal declined again to previous low levels where it remained stable in the later parts of Sub-zone R1b, with its lowest relative abundance (2 particles/$cm^3$) experienced in ca. AD 1950 (52 cm). The relative abundance of micro-charcoal was on average the second highest (mean 1 $cm^2/cm^3$) in the Sub-zone R1b and showed a fluctuating trend that increased throughout the sub-zone. It reached its highest peak (2.6 $cm^2/cm^3$) in ca. AD 1895 (76 cm) and its lowest relative abundance (0.1 $cm^2/cm^3$) in ca. AD 1940 (58 cm).

Although the highest peak for macro-charcoal over the ca. 215-year palaeo-record was in the previous sub-zone, on average the highest relative macro-charcoal abundance was recorded in Sub-zone R2a (ca. AD 1960-2005; 9-45 cm), with a mean of 42 particles/$cm^3$. Initially, the trend showed relatively low variability, with the lowest abundance (2 particles/$cm^3$) in ca. AD 1980 (30 cm), which was the same low abundance recorded in Sub-zone R1b in ca. AD 1950 (52 cm). The trend increased slightly (74 particles/$cm^3$) in the middle of the sub-zone in ca. AD 1990 (24 cm) before decreasing again. Thereafter, there was a dramatic increase to 185 particles/$cm^3$ in ca. AD 2000 (10 cm) at the R2a/R2b transition. Micro-charcoal abundance did not follow the same trend as the macro-charcoal and exhibited its lowest average abundance in R2a (mean of 0.6 $cm^2/cm^3$) compared to other sub-zones. It showed a fluctuating trend that decreased gradually throughout the sub-zone.

In Sub-zone R2b (ca. AD 2005-2011; 9-0 cm), macro-charcoal was on average the second highest (mean 37 particles/$cm^3$) compared to R1a, R1b and R2a. After the high peak at the sub-zone R2a/R2b transition, macro-charcoal abundance decreased to a low level (5 particles/$cm^3$) in ca. AD 2005 (8 cm) and then increased moderately over time towards the later part of R2b. The same trend was seen in the micro-charcoal abundance and although on average, micro-charcoal was the second lowest (mean of 0.9 $cm^2/cm^3$) in R2b compared to other sub-zones, its mean was closer to that of R1a and R1b.

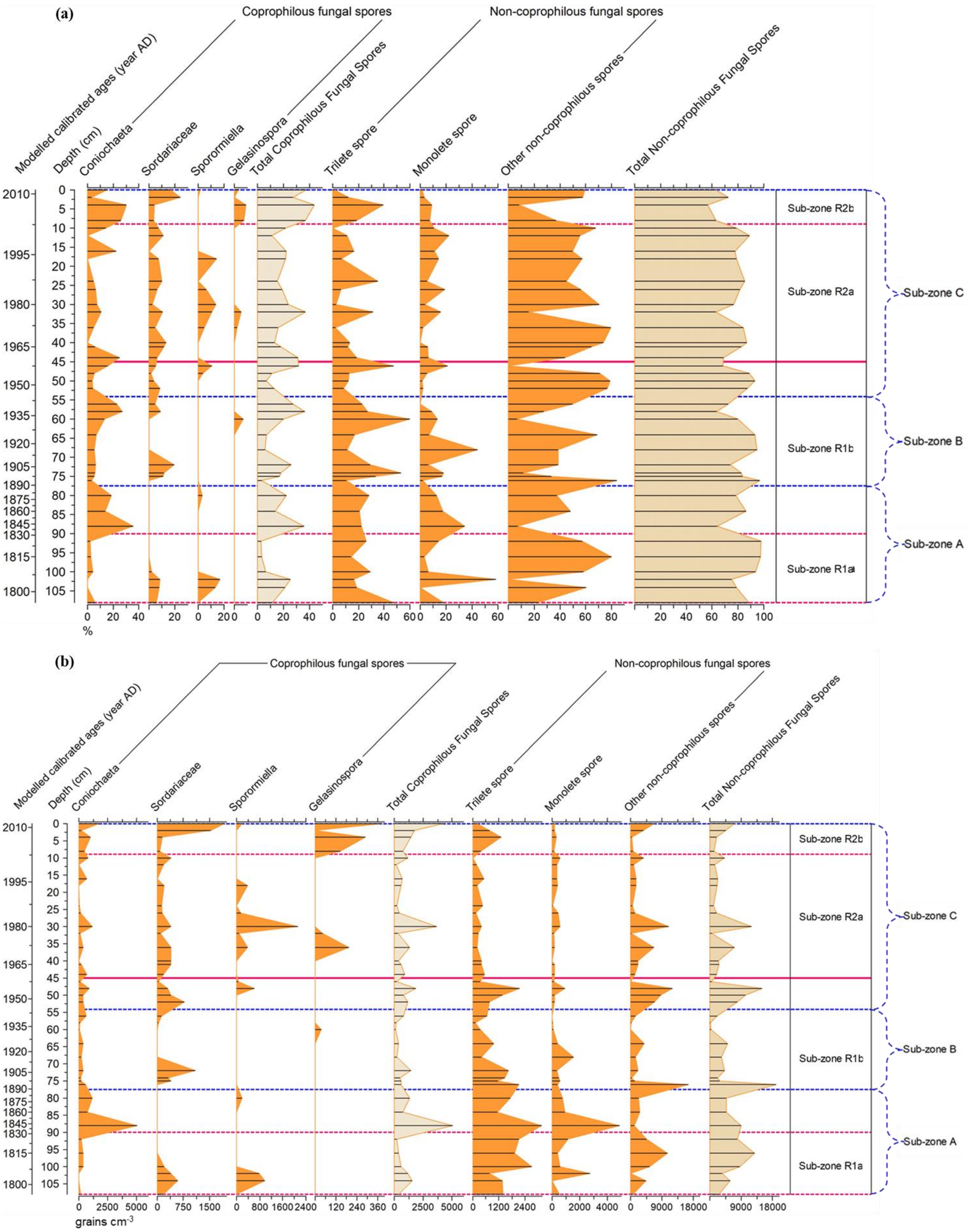



*Figure 6.8: (a) Percentage and (b) concentration spores diagrams for the RV3 core at Rhenostervlei Farm. Spore types are grouped into coprophilous fungal spores (a proxy for herbivory) and non-coprophilous fungal spores. Pollen (pink lines) and diatom (blue lines) zonation depicted on the right and modelled calibrated ages are shown on the far left.*

### 6.5.3 Spores

#### *6.5.3.1 Coprophilous Fungal Spores as a proxy for herbivory*

In Sub-zone R1a (ca. AD 1795-1825; 108-90 cm), the mean of aggregated coprophilous fungal spores (the sum of *Coniochaeta,* Sordariaceae, *Sporormiella* and *Gelasinospora*, as an indicator of relative herbivory abundance) was relatively low (mean 12% and 704 grains/cm$^3$), with a trend of slight fluctuation as it increased initially and then decreased in the later part of the sub-zone. The abundance of individual coprophilous fungal spore taxa, *Coniochaeta,* Sordariaceae and *Sporormiella* increased initially and decreased during the later parts of this sub-zone, whereas *Gelasinospora* was completely absent throughout ca. AD 1795-1825. These trends were apparent in both the percentage and concentration data. However, the lowest abundance of the sum of coprophilous fungal spores was in R1a for the percentage data (2%) in ca. AD 1820 (92 cm), whereas the lowest abundance was in R1b for the concentration data (109 grains/cm$^3$) in ca. AD 1935 (60 cm)).

In Sub-zone R1b (ca. AD 1830-1960; 90-45 cm) the mean of aggregated coprophilous fungal spores was relatively high at 18% and 993 grains/cm$^3$. There was a more pronounced trend in the concentration data compared to the percentage spore data, likely due to the Fagerlind effect. The trend in the concentration of total coprophilous fungal spore data initially showed an unprecedented high relative abundance (5 068 grains/cm$^3$) in ca. AD 1845 (88 cm) in the record from this site, which decreased and fluctuated at a low level to its lowest abundance (109 grains/cm$^3$) in ca. AD 1935 (60 cm). Coprophilous fungal spores then increased slightly before decreasing again in the later part of the sub-zone. These trends were not apparent in the percentage data where coprophilous fungal spores fluctuated with several highs and lows throughout the Sub-zone R1b. Trends in individual coprophilous fungal spore taxa are as follows: the initial trend of the most abundant coprophilous fungal spore taxa, *Coniochaeta* (mean 12%), was unprecedently high for both the percentage and concentration data, 36% and 5 068 grains/cm$^3$, respectively, just after the sub-zone R1a/R1b transition in ca. AD 1845 (88 cm). *Coniochaeta* abundance decreased during the middle part of the sub-zone and in the percentage data it increased and decreased again, whereas in the concentration data it fluctuated at a low level throughout the rest of Sub-zone 1b. The different pattern in the percentage data is likely due to the Fagerlind effect. Sordariaceae was absent from ca. AD 1815 (76 cm) but reappeared in Sub-zone R1b in ca. AD 1895 (96 cm) (ca. 80 years later). It showed a high peak (19% and 1 094 grains/cm$^3$) in ca. AD 1905 (72 cm), was absent from the sub-zone again until ca. AD 1940 (58 cm), when it increased and decreased again towards the later part of the sub-zone. *Sporormiella* was absent in the majority of the sub-sampled depths (27/38), most notably around the sub-zone R1a/R1b transition between ca. AD 18010-1860 (100-84 cm) (for ca. 50 years), with the longest period of absence occurring in R1b between ca. AD 1895-1950 (76-50cm) (ca. 55 years). It showed a low abundance in ca. AD 1880 (80 cm) and at the sub-zone R1b/R2a transition in ca. AD 1955-1960 (48-46cm). *Gelasinospora* was present for the first time at a low abundance (7% and 36 grains/cm$^3$) in ca. AD 1935 (60 cm).

In Sub-zone R2a (ca. AD 1960-2005; 45-9 cm), the mean of aggregated coprophilous fungal spores increased to a relatively higher mean of 21% in the percentage data but decreased to a lower mean of 866 grains/cm$^3$ in the concentration data. The conflicting trends are likely due to the Fagerlind effect in the percentage data. The trends in the percentage and concentration data were similar in that they fluctuated throughout with a noticeable peak that occurred towards the middle of the sub-zone: 37% 3 673 grains/cm$^3$ in ca. AD 1980 (32-30 cm). Trends of individual coprophilous fungal spore taxa are as follows: *Coniochaeta* generally decreased (mean 8% and 349 grains/cm$^3$) in Sub-zone R2a, with a trend that fluctuated at a relatively low level throughout, reaching its lowest abundance for the concentration data only (65 grains/cm$^3$) in ca. AD 1990 (24 cm). Sordariaceae generally increased in abundance (mean 7.54% and 237 grains/cm$^3$). It fluctuated and decreased slightly towards the later part of the sub-zone when it was absent in ca. AD 1995 (16 cm) before increasing slightly at the R2a/R2b transition. *Sporormiella* was more abundant during R2a (mean 4.1% and 261 grains/cm$^3$) compared to the previous sub-zone, with a significantly high peak only evident in the concentration data (2 127 grains/cm$^3$) in ca. AD 1980 (30 cm). *Gelasinospora* remained low in abundance (mean 0.63% and 20 grains/cm$^3$) and was only present in ca. AD 1970 (36 cm) and ca. AD 1980 (32 cm).

In Sub-zone R2b (ca. AD 2000-2011; 0-9 cm), as an indicator of relative herbivory abundance, the aggregated coprophilous fungal spores were at their highest mean for both the percentage (mean 36%) and concentration (mean 2 039 grains/cm$^3$) data. However, the trends in the percentage and concentration data varied. In the percentage data it increased until ca. AD 2010 (4 cm), decreased and then increased again at the later part of the sub-zone. Whereas in the concentration data, total coprophilous fungal spores increased steeply over time to a high relative abundance of 4 253 grains/cm$^3$ in ca. AD 2011 (0 cm). Trends of individual coprophilous fungal spore taxa showed that *Coniochaeta* generally increased (mean 19% and 862 grains/cm$^3$) in relative abundance, with a trend that fluctuated over time, with a low trough in ca. AD 2010 (2 cm) evident in both the percentage (3%) and concentration data (193 grains/cm$^3$) and an increase in ca. AD 2011. On average, Sordariaceae abundance was the highest throughout the RV3 palaeo-record in R2b for both the percentage data and the concentration data (mean 12.31% and 924 grains/cm$^3$). Percentage Sordariaceae data increased to its highest peak (24%) in ca. AD 2010 (2 cm) and then decreased again, whereas in the concentration data the trend continued to increase until ca. AD 2011 (0 cm) when it experienced an unprecedented high (1 933 grains/cm$^3$) in the record from this site. *Sporormiella* was mostly absent during R2b (mean 0.42% and 48 grains/cm$^3$) with low abundance during the latter part of Sub-zone R2b in ca. AD 2011. *Gelasinospora* abundance increased significantly (mean 4.87% and 205 grains/cm$^3$). It was its highest for the percentage data (9%) in ca. AD 2108 (4 cm), and the highest for concentration data (387 grains/cm$^3$) in ca. AD 2011 (0 cm). The *Gelasinospora* trend showed an increase, then a decrease until ca. AD 2010 (2 cm) where it was absent before increasing again.

#### *6.5.3.2 Non-Coprophilous Spores*

In Sub-zone R1a (ca. AD 1795-1825; 108-90 cm) the mean of aggregated non-coprophilous spores (the sum of Trilete, Monolete and other non-coprophilous spores) showed a relatively high mean of 88% and 6 915 grains/cm$^3$. The trend showed slight fluctuation as it increased and then decreased during the sub-zone. Other non-coprophilous fungal spores (mean 47% and 4 305 grains/cm$^3$) were most abundant, followed by Trilete spores (mean 25% and 1 715 grains/cm$^3$) and Monolete spores (mean 17% and 896 grains/cm$^3$). Trends showed high variability throughout the sub-zone and were consistent between percentage and concentration spore data. However, the highest total of non-coprophilous spores was in R1a for the percentage data (98%) in ca. AD 1820 (92 cm) just before the sub-zonal R1a/R1b transition, whereas the highest concentration data (19 139 grains/cm$^3$) relative abundance was during R1b in ca. AD 1895 (76 cm). This difference is a result of the Fagerlind effect.

In Sub-zone R1b (ca. AD 1830-1960; 90-45 cm), non-coprophilous spore abundance was relatively lower on average (mean of 82% and 5 757 grains/cm$^3$). Interestingly, both Trilete (3 168 grains/cm$^3$) and Monolete data (4 857 grains/cm$^3$) spores showed an unprecedented high peak in the concentration data just after the sub-zone R1a/R1b transition in ca. AD 1845 (88 cm), which was not evident in the percentage data likely due to the Fagerlind effect. Initially there was a decline in total non-coprophilous spore abundance which then increased dramatically (97% and 19 139 grains/cm$^3$) in ca. AD 1895 (76 cm)). It then decreased to a very low abundance (64% and 254 grains/cm$^3$) in ca. AD 1940 (58 cm), before increasing again (89-93 % and 10 246-15 036 grains/cm$^3$) in ca. AD 1950-1955 (50-48 cm) in the later parts of the sub-zone and decreasing again (68% and 479 grains/cm$^3$) just before the R1b/R2a transition in ca. AD 1960 (46 cm).

In Sub-zone R2a (ca. AD 1960-2005; 45-9 cm), patterns in the aggregated non-coprophilous spore abundance decreased on average (mean of 79% and 3 327 grains/cm$^3$) compared to the previous sub-zone. In the percentage data, the trend continued to fluctuate at a similar level as before whereas in the concentration there was a slight increasing trend over time with a noticeable peak (11 986 grains/cm$^3$) in ca. AD 1980 (30 cm). Trilete spores were relatively lower in abundance in R2a (mean of 13% and 301 grains/cm$^3$). The trend in the percentage trilete spore data was very variable, with multiple high peaks and troughs, whereas the trend in the concentration data showed fluctuation at a lower level. Trilete spores were only absent from the RV3 palaeo-record once and this was just before the sub-zone R2a/R2b transition in ca. AD 2000 (10 cm). Monolete spore abundance also decreased on average in R2a (mean of 10% and 285 grains/cm$^3$). Monolete spores fluctuated with a slightly increasing trend over time.  Other non-coprophilous spores increased in the early parts of the sub-zone (11 019 grains/cm$^3$) in ca. AD 1980 (30 cm), then decreased and was relatively stable at a lower level.

On average, in Sub-zone R2b (ca. AD 2000-2011; 9-0 cm), non-coprophilous spore abundance decreased in the percentage data (mean of 64%) but increased slightly in the concentration data (mean 3 724 grains/cm$^3$) compared to the previous sub-zone. Like the trends of the coprophilous fungal spores, there was variation in trends in percentage and concentration data. Percentage total non-coprophilous decreased until ca. AD 2010 (4 cm), increased slightly and then decreased again at the later part of the sub-zone. Whereas in the concentration data, non-coprophilous fungal spores increased moderately over time. Trilete and monolete spores showed similar trends as they increased until ca. AD 2010 (4 cm) and then decreased again. More noticeable in the concentration data than the percentage data, trilete spores had a peak 1 302 grains/cm$^3$) in abundance that was not evident in the previous Sub-zone R2a but did resemble the means of previous sub-zones, R1a and R1b. Other non-coprophilous spores experienced its lowest mean of 41% and 2 862 grains/cm$^3$ in R2b. In contrast to the trends of trilete and monolete spores, other non-coprophilous spores decreased to a low level in ca. AD 2010 (4 cm) and increased again in the later parts of the sub-zone.

### 6.5.4 Diatom results for RV3 core: A proxy for water quality and climate reconstruction

#### *6.5.4.1 Diatom zonation*

Table 6.5 summarises the diatom zonation results (dashed lines for Sub-zone A (108-78 cm), Sub-zone B (78-54 cm) and Sub-zone C (54-0 cm)) illustrated in Figure 10.18, which depicts the distribution of percentage diatom taxa based on depth of the RV3 core. Note that this cluster analysis was performed separately to that of the fossil pollen assemblage to isolate the timing of changes in multi-proxy palaeo data for ecosystem services (see section 4.1.4.2). The broken stick model (Figure 10.17) indicated that there were no significant zones because the value of the black line was not above the critical value represented by the red line. Despite a lack of statistical significance in the diatom zonation, the depth factor was divided into three sub-zones to describe changes in the diatom assemblage based on the square root transformed diatom data and the Bray-Curtis similarity data.

***Table 6.5: Details of RV3 diatom zonation. Dashed lines indicate sub-zones that are not statistically significant.***

| Diatom Sub-zone | Number of sub-sample depths analysed | Depth ranges of samples (cm) | Dates of diatoms zone boundaries (AD) | Number of years |
|---|---|---|---|---|
| C | 20 | 0-54 | 1945-2011 | 65 |
| B | 9 | 54-78 | 1890-1945 | 55 |
| A | 9 | 78-108 | 1795-1890 | 95 |

Diatom results are categorised according to autecology (habitat, salinity and nutrient preferences) (Gell et al., 2007; Kirsten, 2014; Stager et al., 2013, 2012; Taylor et al., 2007) (see Table 4.2 in section 4.1.3.3). Results are summarised in Figure 6.9 with detailed results illustrated in the Appendix according to diatoms habitat (Figure 10.19), salinity (Figure 10.20) and nutrients (Figure 10.21) preferences.

#### *6.5.4.2 Diatom Sub-Zone A (9 sub-sample depths; 108-78 cm; ca. AD 1795-1890)*

Sub-zone A, representing depths 108-78cm (ca. AD 1795-1890), is characterized by diatom taxa that inhabited substrates that experienced standing/open water (benthic) with an interplay of episodic drying out (aerophilic) (Figure 6.9). Such conditions are also characteristic of brackish-saline pool (high salinity), particularly during ca. AD 1795-1805 (108-102cm). *Nitzschia filiformis* (Appendix Figure 10.20) was relatively high in relative abundance (21%) in ca. AD 1795 (108 cm). *Nitzschia filiformis* is tolerant of strongly polluted conditions, whether human-induced or due to surface run-off from organic matter at the site, but not tolerant of critical brackish-saline water conditions, and thus survives in medium levels of salinity (brackish). Nutrient enhancement at the site may not be the driving mechanism at the time but rather the trend toward these conditions. *Gyrosigma acuminatum* peaked at (20%) in ca. AD 1805 (102 cm). This species can tolerate critical levels of organic pollution and higher saline conditions. *Tryblionella coarctata* is another indicator diatom species with a preference for brackish-saline water quality (Appendix Figure 10.20). Within Sub-zone A, depths 100-96cm (ca. AD 1810-1815) experienced low diatom preservation and therefore could not be analysed. During ca. AD 1820-1880 (92-80cm) a slight recovery period occurred as *Hantzschia amphioxys* and

*Pinnularia borealis* decreased slightly, and freshwater and (tyco)planktonic species (*Aulacoseira granulata (+ var. angustissima)*) increased (Appendix Figure 10.19). The water quality of the wetland was fresh to fresh-brackish and experienced damp/moist ground that supported a benthic community over an aerophilic community (Appendix Figure 10.20).

#### *6.5.4.3 Diatom Sub-Zone B (9 sub-sample depths; 54-78 cm; ca. AD 1890-1945)*

Sub-zone B, representing depths 78-54 cm (ca. AD 1890-1945), showed an increase in diatom taxa representing a benthic community with a water quality preference characterised by standing or open water and medium salinity (Figure 6.9). The most abundant taxa, *Nitzschia filiformis*, peaked (26%) during ca. AD 1920 AD (68 cm) (Appendix Figure 10.20). Prior to this there is a notable increase in *Staurosira elliptica* (18% in ca. AD 1905 (72 cm)) which is a species that prefers benthic (standing/open water) environments (Appendix Figure 10.19) that are fresh-brackish (low salinity) (Appendix Figure 10.20) with medium nutrient availability (Appendix Figure 10.21). *Staurosira elliptica* is a typical pioneer species. A slight peak in *Gyrosigma acuminatum* (10%) is evident at the beginning of Sub-zone B in ca. AD 1895-1900 (76-75 cm).

#### *6.5.4.4 Diatom Sub-Zone C (20 sub-sample depths; 54-0 cm; ca. AD 1945-2011)*

Sub-zone C, representing depths 54-0 cm (ca. AD 1945-2011), experienced episodic drying out and wetter periods (Figure 6.9). The beginning of Sub-zone C is characterised by high freshwater inputs at ca. AD 1945 AD (54 cm) which steadily declines and becomes brackish by ca. AD 19780 (32 cm). *Aulacoseira granulata (+ var. angustissima)* remains prevalent throughout the sub-zone and peaked (33% and 29%) in ca. AD 1950 (52 cm) and ca. AD 1980 (32 cm) (Appendix Figure 10.19). Another (tyco)planktonic species, *Melosira varians*, also peaked at the beginning of Sub-zone C, 13% and 11% in ca. AD 1950 and 1960, respectively (50 cm and 44 cm) (Appendix Figure 10.19). *Melosira varians* prefers high nutrient availability (Appendix Figure 10.21), circumneutral water (pH ±7), and is easily suspended into the water column. From ca. AD 1980-2011 (0-31 cm) there are further changes in water chemistry. *Cyclotella meneghiniana* peaked (11%) in ca. AD 1995 (16 cm). This is a (tyco)planktonic species that is easily suspended into the water column (indication of regional or wetter conditions) (Appendix Figure 10.19), occurs in fresh-brackish (Appendix Figure 10.20) and high nutrient water quality (Appendix Figure 10.21). *Achnanthidium minutissimum*, a benthic species which often occurs in oxygenated, clean, fresh waters and is often associated with physical disturbance, peaked (9%) in ca. AD 2010 (4 cm) (Appendix Figure 10.19).

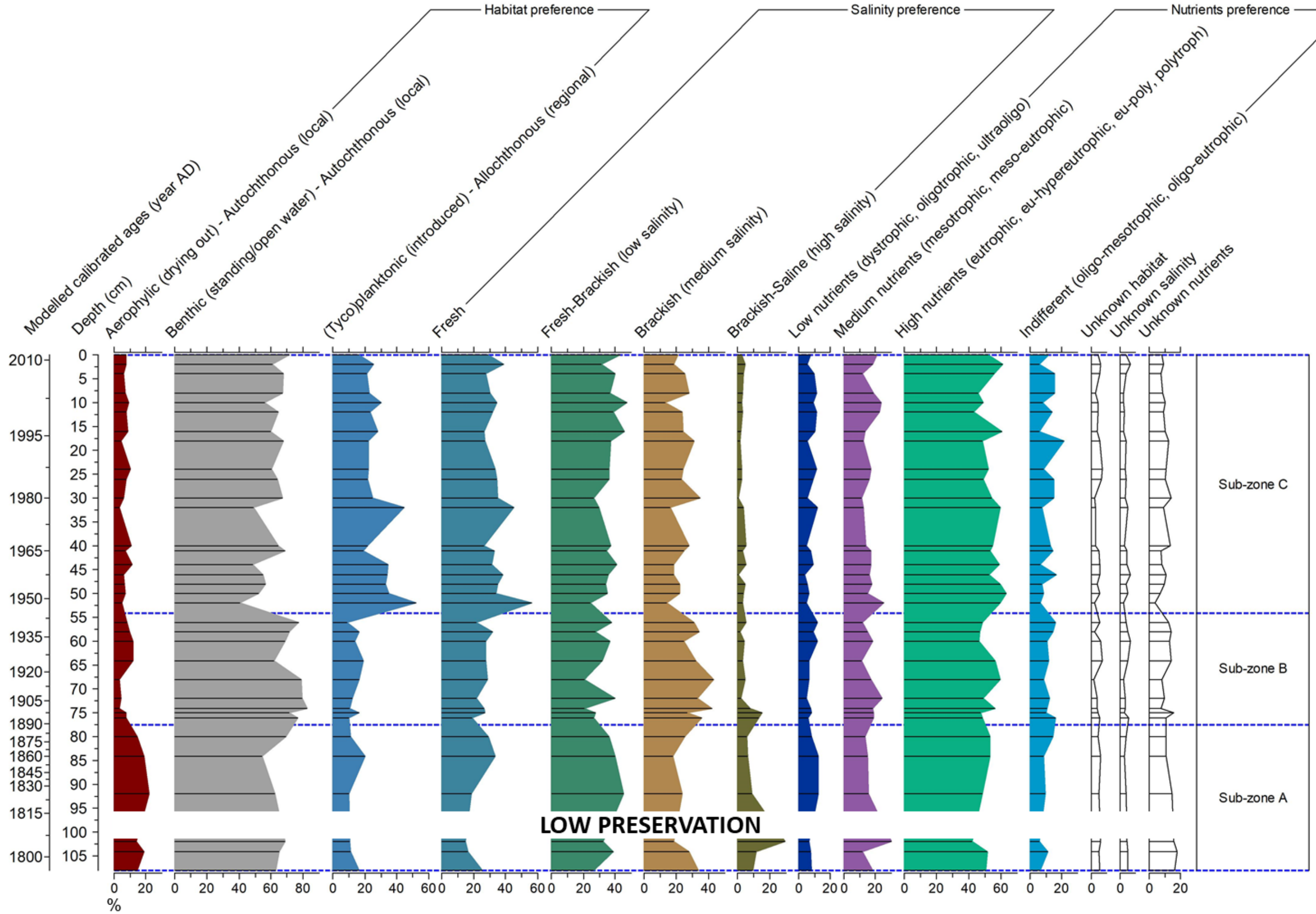



*Figure 6.9: Summary diagram showing percentage diatom data for RV3 core. Categories include sums of diatom taxa with various habitat, salinity and nutrients preferences. See Appendices Figure 10.19, Figure 10.20 and Figure 10.21 for the full unabridged species lists. Low diatom preservation is shown between 100-96 cm in Sub-zone A.*

### 6.5.5 Geochemical (XRF) and physical properties (MS and LoI)

In Sub-zone R1a (ca. AD 1795-1825; 108-90 cm), on average Rb:K (mean $5.48E^{-3}$) and Ba (mean $4.27E^{-2}$ %) were their highest whereas Zr:Rb (mean ratio 3.6) experienced its lowest mean in R1a compared to other sub-zones. Like the trend of volume MS, elements Fe, Mn, Ca and Zr:Rb (erosion) increased initially, decreased during the middle part of the sub-zone and then increased slightly towards the later part. The general trend of K, Ba, Cu, Mg, P, Zn, Cs and Rb:K (salinity ratio) showed a gradual increase throughout Sub-zone R1a. Fe (7%) and Mn ($3.01E^{-1}$ %) both increased to their highest abundance in ca. AD 1800 (104 cm). Interestingly, Ca ($2.21E^{-1}$ %) also experienced a high peak at ca. AD 1800 (104 cm), whereas Cs ($4E^{-4}$ %) and P (0.025%) experienced their lowest abundance at this time. Furthermore, K (2.3%) and Cu ($2.56E^{-3}$ %) reached their highest relative abundance just before the sub-zone R1a/R1b transition in ca. AD 1820 (92 cm), whereas Zr:Rb (2.1) was its lowest at this time. Both low-frequency and volume MS (mean $5.32E^{-2}$ $\chi$lf and $7.13E^{-5}$ SI, respectively) were second lowest in R1a compared to the average relative abundance of MS in R1b, R2a and R2b. Volume MS increased in the early part of the sub-zone and then fluctuated at a relatively stable level. $LoI_{550}$ (organic carbon) was its highest (mean 5.4%) in Sub-zone R1a, which increased to its highest relative abundance (7.6%) in ca. AD 1800 (104 cm), then decreased slightly towards the later part of the sub-zone. Inorganic carbon was relatively high initially but then decreased significantly (0.1%) in ca. AD 1805 (102 cm), increased thereafter and decreases slightly in the later parts. On average, water content was highest (mean 17.2%) in Sub-zone R1a. It showed a trend that increased initially and then fluctuated at a stable high abundance, with its highest peak (20.31%) in ca. AD 1820 (92 cm) before the R1a/R1b transition.

In Sub-zone R1b (ca. AD 1830-1960; 90-45 cm), the general trend of elemental K, Ba, Cu, Mg, Zn, Fe, Mn, Ca, P and Rb:K (salinity ratio) showed an initial slight decrease followed by a gradual increase towards the middle parts of the sub-zone in ca. AD 1895-1900 (76-74cm). Interestingly, Rb:K (salinity ratio) was its highest ($6.35E^{-3}$) in Sub-zone R1b in ca. AD 1900 (75 cm) and Ba (algal blooms) reached its highest ($5.43E^{-2}$ %) in ca. AD 1895 (76 cm). The trend in these elemental properties abundance then decreased to an unprecedented low abundance in ca. AD 1940 (58 cm) in the record from this site: K (0.7%), Ba ($1.7E^{-2}$ %), Cu ($4.9E^{-4}$ %), Mg ($2E^{-3}$ %), Zn ($1.1E^{-3}$ %), Fe (0.6%), Mn ($7.7E^{-3}$ %), Ca ($1E^{-3}$ %). Although not unprecedented for P abundance (0.04%) but it also decreased around this time (ca. AD 1940 (56 cm)). Their trends increased again in ca. AD 1950 (52-50 cm) to previous high levels before decreasing slightly at the R1b/R2a transition. Ca and Zr:Rb (erosion ratio) showed a different trend compared to the other elements where their abundance decreased initially. .

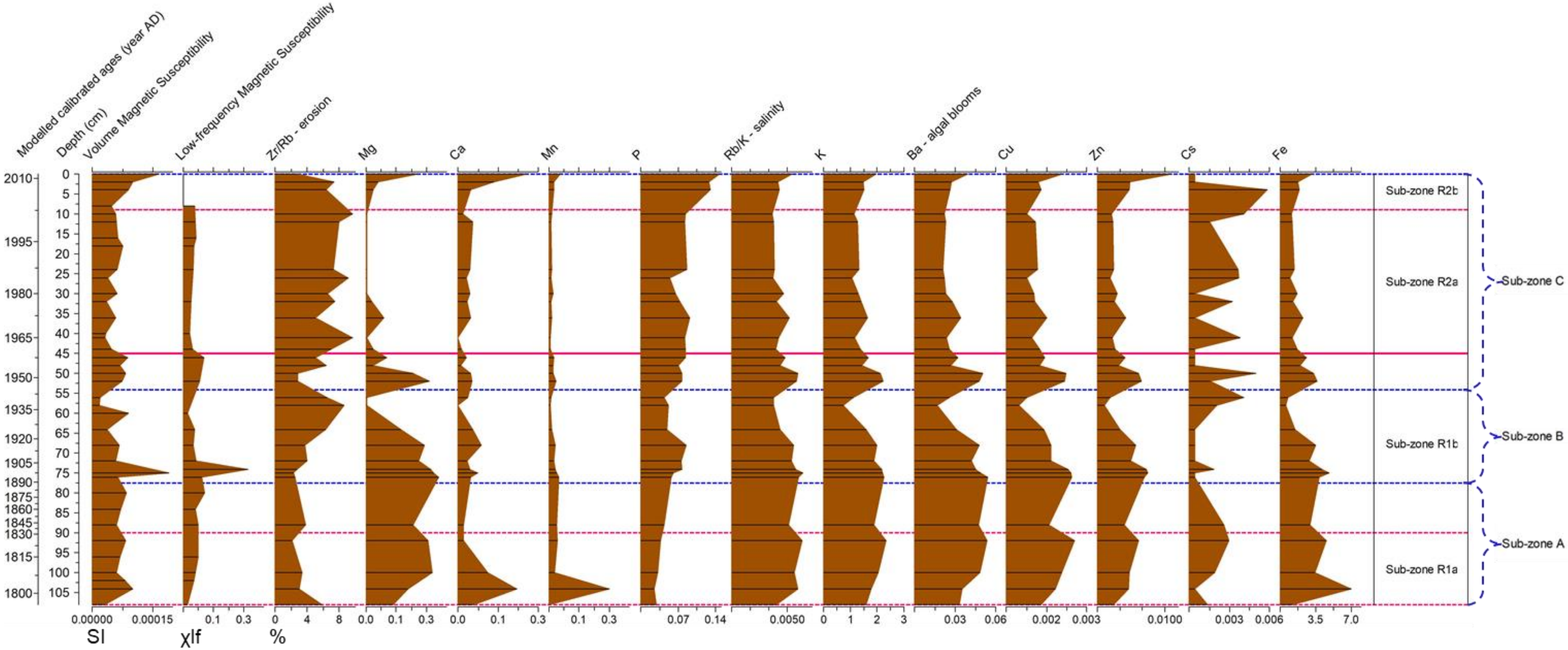



*Figure 6.10: Physical properties (Magnetic Susceptibility) and geochemical (elemental composition) diagrams for the RV3 core as a proxy for soil regulation at Rhenostervlei Farm. Pollen (pink) and diatom (blue) zonation depicted on the right and modelled calibrated ages are shown on the far left.*

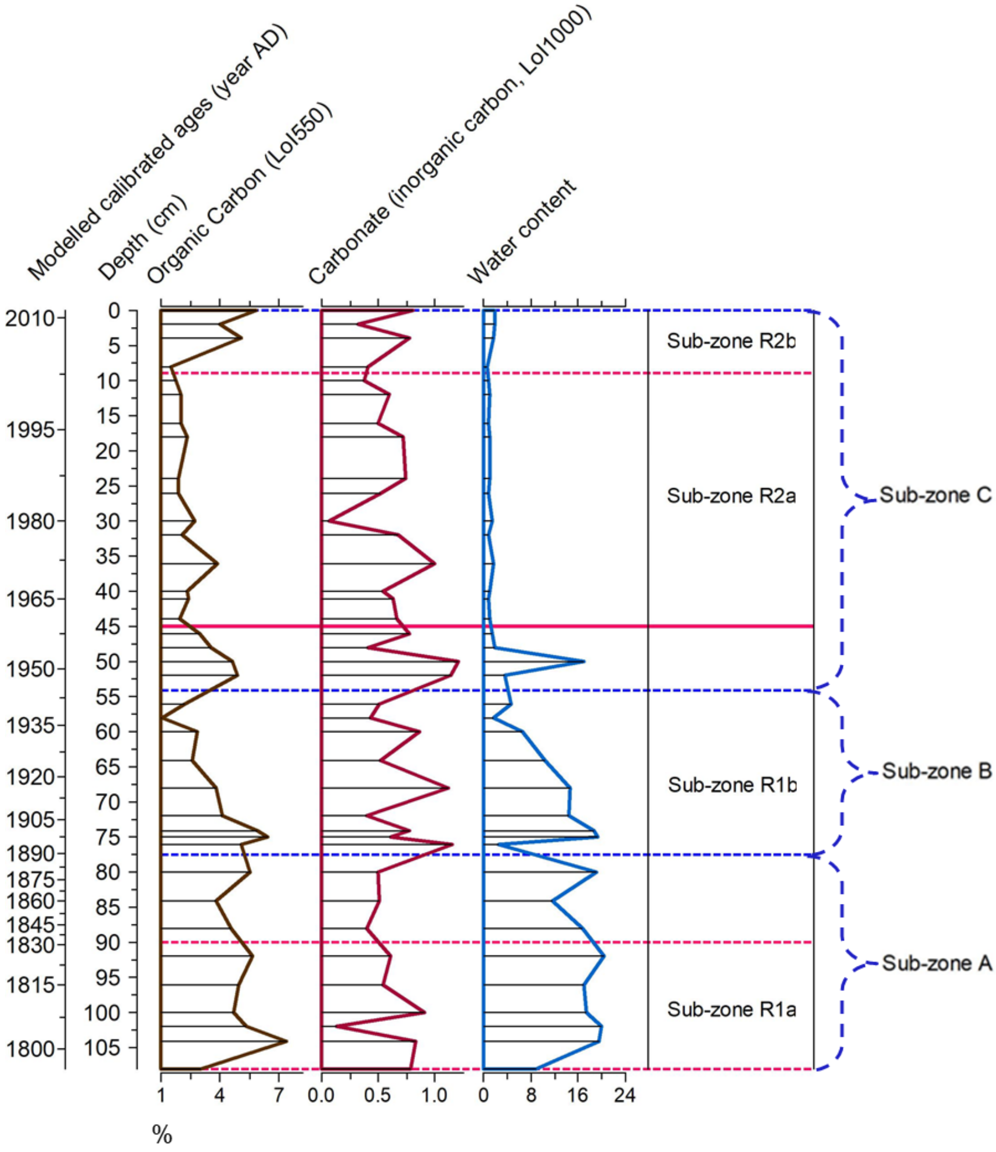



***Figure 6.11: Percentage of organic carbon ($LoI_{550}$) and carbonate ($LoI_{1000}$, Inorganic carbon, $CO_3$) content per dry weight, and percentage water content per wet weight of the RV3 core sediments. Zonation of pollen data (pink) and diatom data (blue) depicted on the right and modelled calibrated ages are shown on the far left.***

Thereafter, their trends increased ($2.06E^{-3}$ % for Cs and a ratio of 9 for Zr:Rb erosion ratio) in ca. AD 1940 (58 cm) when other elements decreased at this time. Interestingly, Cs did experience peaks like the other elements: $4.11E^{-3}$ % in ca. AD 1900 (74 cm) and $4.97E^{-3}$ % in ca. AD 1950 (50 cm). Furthermore, on average, Rb:K (mean $5.48E^{-3}$ salinity ratio) and Ba (mean $4.27E^{-2}$ %) were the second highest means whereas Zr:Rb (mean 3.6 erosion ratio) was the second lowest in R1b compared to the other sub-zones.

On average, the relative abundance of low-frequency MS was the highest (mean $9.4E^{-2}$ $\chi$lf) in R1b whereas volume MS was the second highest mean (mean $7.17E^{-5}$ SI) in this sub-zone compared to the average relative abundance of R1b, R2a and R2b. Volume MS increased in the early part of the sub-zone and then fluctuated at a relatively stable level. General trends for the low-frequency MS and volume MS were similar. Volume MS reached a dramatically high peak ($1.91E^{-4}$ SI) in ca. AD 1900 (75 cm) and the highest low-frequency MS peak ($3.23E^{-1}$ $\chi$lf) was shortly after that in ca. AD 1900 (74 cm). The trend then decreased in the later parts of Sub-zone R1b with low-frequency MS reaching its lowest abundance ($2.19E^{-2}$ $\chi$lf) in ca. AD 1935 (60 cm) and volume MS experienced its lowest abundance ($1.86E^{-5}$ SI) in ca. AD 1940 (58 cm). The trend increased again at the R1b/R2a transition. On average, $LoI_{550}$ (organic carbon) was its second lowest (mean 4.2%) in Sub-zone R1b. $LoI_{550}$ abundance decreased initially, then increased (6.6%) in ca. AD 1900 (75 cm), before decreasing gradually to its lowest level (1.3%) in ca. AD 1940 (cm 58). The trend increased again and then decreased in the later parts of the sub-zone. Inorganic carbon showed a similar trend to organic carbon initially and increased to a high peak (1.15%) in ca. AD 1895 (cm 76). Inorganic carbon fluctuated in the middle parts of Sub-zone R1b with the highest peak (1.21%) in ca. AD 1950 (50 cm). On average, water content was second highest (mean 10.3%) in Sub-zone R1b. It showed a trend that decreased, increased, and then decreased again dramatically to 2.45% in ca. AD 1895 (76 cm), before increasing again (19.25%) in ca. AD 1900 (75 cm). During the second part of the sub-zone, water content decreased gradually to 1.76% in ca AD 1940 (58 cm) before increasing to a higher relative abundance (17.07%) in ca. AD 1950 (50 cm) and decreasing again in the later part of Sub-zone R1b.

In Sub-zone R2a (ca. AD 1960-2005; 45-9 cm), geochemical analyses yielded the following results: Mg, Ca and Cs reached their lowest relative abundance. Rb:K (salinity ration) experienced its lowest mean (mean $4.1E^{-3}$) in R2a and also reached its lowest relative abundance ($3.65E^{-3}$) in ca. AD 2000 (10 cm). Conversely, the highest Zr:Rb (erosion ratio 9.7) was in R2a in ca. AD 1965 (41 cm), and on average the Zr:Rb was highest (mean ratio 7.7) during this sub-zone compared to the other sub-zones of the RV3 record. Ba (algal blooms) was on average the lowest mean of $2.48E^{-2}$ % in R2a. The general trend of K, Ba, Cu, Zn, Fe, Ca, P and Rb:K (salinity ratio) showed an initial decrease followed by a slight increase in the early parts of the sub-zone with a peak in ca. AD 1970 (36 cm). Their abundances remained stable with a slight decline at R2a/R2b transition. P showed a slight variation in this trend where it decreased to 0.05% in ca. AD 1985 (26 cm) and then increased during the latter part of the sub-zone. Although Mg followed an initial trend similar to elements described above, it was at undetectable levels in the remaining parts of the sub-zone until the R2a/R2b transition and showed significantly lower average abundance in Sub-zone R2a (mean $1.22E^{-2}$ %). Mn experienced its lowest mean of $1.22E^{-2}$ % in R2a, with a trend that was stable at a low level throughout the sub-zone. The Cs elemental abundance was highest in R2a (mean $2.36E^{-3}$ %) and the trend was extremely variable throughout Sub-zone R2a, with multiple high peaks in ca. AD 1965 (41 cm), ca. AD 1980 (32 cm), ca. AD 1985 (26 cm) and ca. AD 2000 (10 cm); and low troughs in ca. AD 1970 (36 cm) and ca. AD 1980 (30 cm)). Relative abundance of both the low-frequency MS (mean $4.95E^{-2}$ $\chi$lf) and volume MS (mean $5.24E^{-5}$ SI) was the lowest in R2a compared to other sub-zones. MS decreased in the early part of the sub-zone and then fluctuated at a relatively stable level with a slight increase over time. Moreover, on average, $LoI_{550}$ (organic carbon) was its lowest in Sub-zone R2a (mean 2.5%) as it initially increased slightly then decreased and remained stable at a relatively low abundance. Initially, inorganic carbon decreased slightly and then increased to a high peak (1%) in ca. AD 1970 (36 cm). It decreased to its lowest carbonate content (0.1%) in ca. AD 1980 (30 cm) and then increased again before decreasing slowly towards the later parts of the sub-zone. Water content was its lowest (mean 1.1%) in Sub-zone R2a and remained stable at a low level.

In Sub-zone R2b (ca. AD 2000-2011; 9-0 cm), geochemical analyses of the following elements showed a similar trend to MS as they increased over time during Sub-zone R2b. Rb:K (salinity ratio), K, Ba, Fe and Mn increased more subtly whereas Cu, Zn, Ca and P showed relatively dramatic increases from early parts to later parts of the sub-zone with some unprecedently high relative abundances in the record from this site. Zn reached its highest ($1.1E^{-2}$ %) in R2b in ca. AD 2011 (0 cm), which was 2.6 times higher than the average quantity of Zn. Ca ($2.58E^{-1}$ %) and P (0.15%) also reached their highest relative abundances in ca. AD 2011 (0 cm). Cs showed a different trend as it increased to its highest quantity ($5.85E^{-3}$ %) until ca. AD 2010 (4 cm) and then decreased in the later parts of Sub-zone R2b. Interestingly, Cs reached both its highest and lowest ($4E^{-4}$ % relative abundance in ca. AD 2010-2011 (2-0 cm)). On average, Zr:Rb (erosion ratio) was second highest (mean ratio 5.5) in R2b and it showed a decreasing trend throughout the sub-zone, whereas both Rb:K (salinity ration) (mean $4.61E^{-3}$) and Ba (mean $3.12E^{-2}$ %) experienced their second lowest averages in R2b and showed increasing trends. MS analyses showed a similar trend to the geochemical changes as they increased over time during Sub-zone R2b. Typically, the relative abundance of volume MS was highest (mean $1.01E^{-4}$ SI) in R2b compared to the other sub-zones but this was not the case for the average low-frequency MS (mean $5.64E^{-2}$ $\chi$lf) which was the second highest average during this sub-zone. The trend in volume MS decreased slightly in the early part of R2b and then increased steeply over time to a similarly high level ($1.66E^{-4}$ SI) in ca. AD 2011 (0 cm) that was also experienced in R1b ($1.91E^{-4}$ SI) in ca. AD 1900 (75 cm)). $LoI_{550}$ (organic carbon) abundance (mean 4.3%) was like Sub-zone R1b although it showed an increasing trend until near present. Inorganic carbon generally followed the same trend as organic carbon. Water content remained at a stable low level (mean 1.5%) in Sub-zone R2a and experienced its lowest content (0.65%) after the R2a/R2b transition in ca. AD 2005 (8 cm).

### 6.5.6 NMDS Ordination results for selected palaeoenvironmental proxies

The NMDS ordination results illustrates the relationship between pollen percentage data and selected palaeo-proxies as explanatory variables (Figure 6.12). The stress value was 0.102 (Figure 10.22) and the stress plot depicted minimal scatter around the line (Figure 10.23). The original dissimilarities were well preserved in the number of dimensions (linear fit $R^2 = 0.904$). Consequently, the ordination is of satisfactory quality and the similarity relationships between the depths were represented accurately. Pollen zonation analysis (see section 6.5.1.1) determined that two distinct zones were statistically different: depths 45-108 cm of Zone R1 were mostly clustered together whilst depths 0-45 cm of Zone R2 clustered together. The projections of points (pollen taxa and sample depths) onto vectors showed maximum correlation with corresponding environmental variables. The grouping of Zone R2 was associated with an increase in the herbivory gradient as well as an increase in the abundance of Poaceae. Zone 2 was also associated with increased P and an increase in *Eucalyptus* pollen. Other explanatory variables associated with upper depths included: Zr:Rb (erosion ratio), (tyco)planktonic diatom taxa, macro-charcoal and Magnetic Susceptibility (MS). In contrast, the grouping of lower depths (Zone R1) showed that the abundance of *Acacia, Anthospermum* and Oleaceae type-2 pollen increased as the abundance of aerophilic diatom taxa increased. The abundance of *Maytenus oleoides* increased as the gradient of micro-charcoal, Mg and benthic diatom taxa abundance increased.

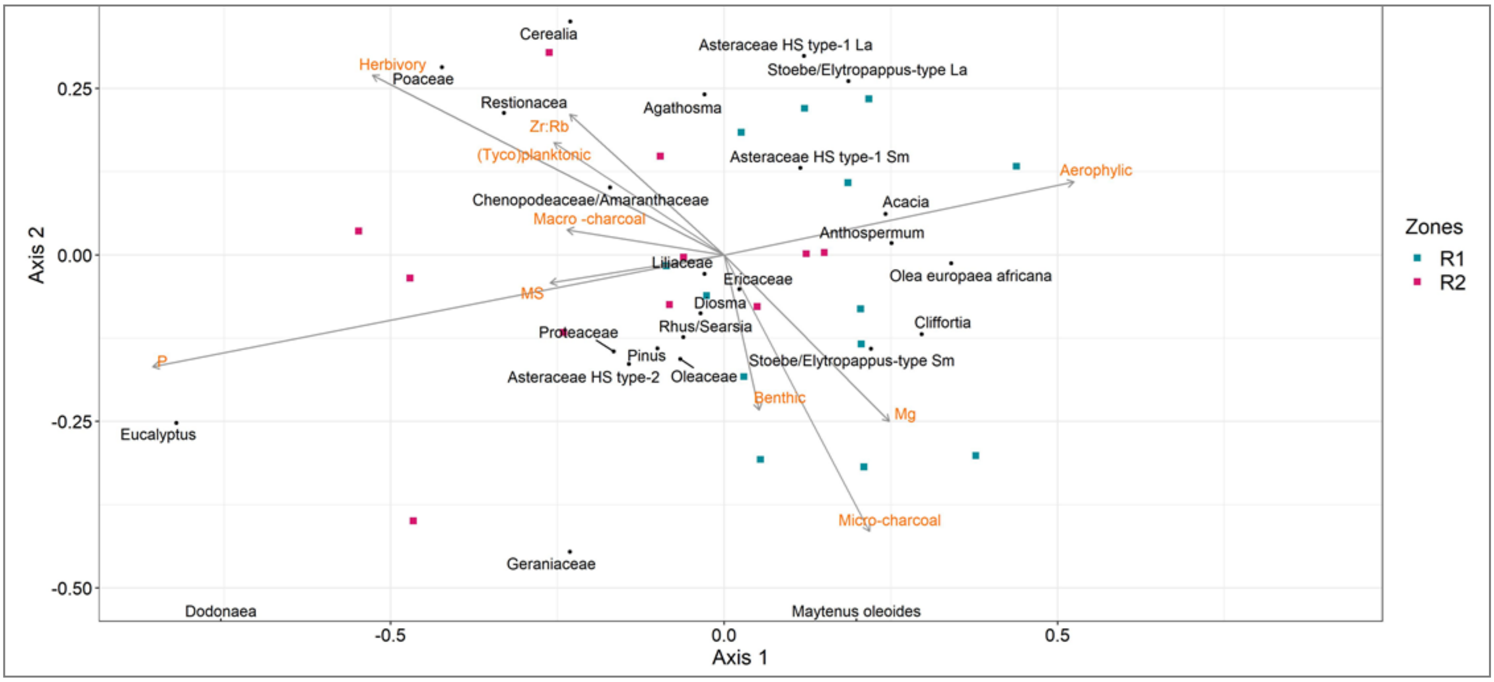



*Figure 6.12: NMDS ordination showing the distribution of pollen taxa based on depth for the RV3 core from Rhenostervlei Farm. The depth factor was classified into two different zones: R1 (shown in pink) and R2 (shown in turquoise). The ordination is based on square root transformed and Bray-Curtis similarity data with a stress factor of 0.102.*

## 6.6 Discussion: Using palaeo-proxies to understand the changes in ecological dynamics at Rhenostervlei Farm

Synthesis of selected multi-proxy palaeoenvironmental data from RV3 core is summarised in Figure 6.13, and patterns in palaeo-proxies and landscape processes are explained below. Furthermore, Figure 6.14 shows the multiple palaeo-proxies from the ca. 215-year-old RV3-record grouped into ecosystem services. Figure 6.15 shows cumulative patterns of changes in indigenous plant biodiversity, which is the essential provisioning and supporting ecosystem services within the Cape Floristic Region (CFR), and how these patterns relate to changes in exotic pollen. In that section, changes in ecosystem services are described and interpreted with reference to resilience theory.

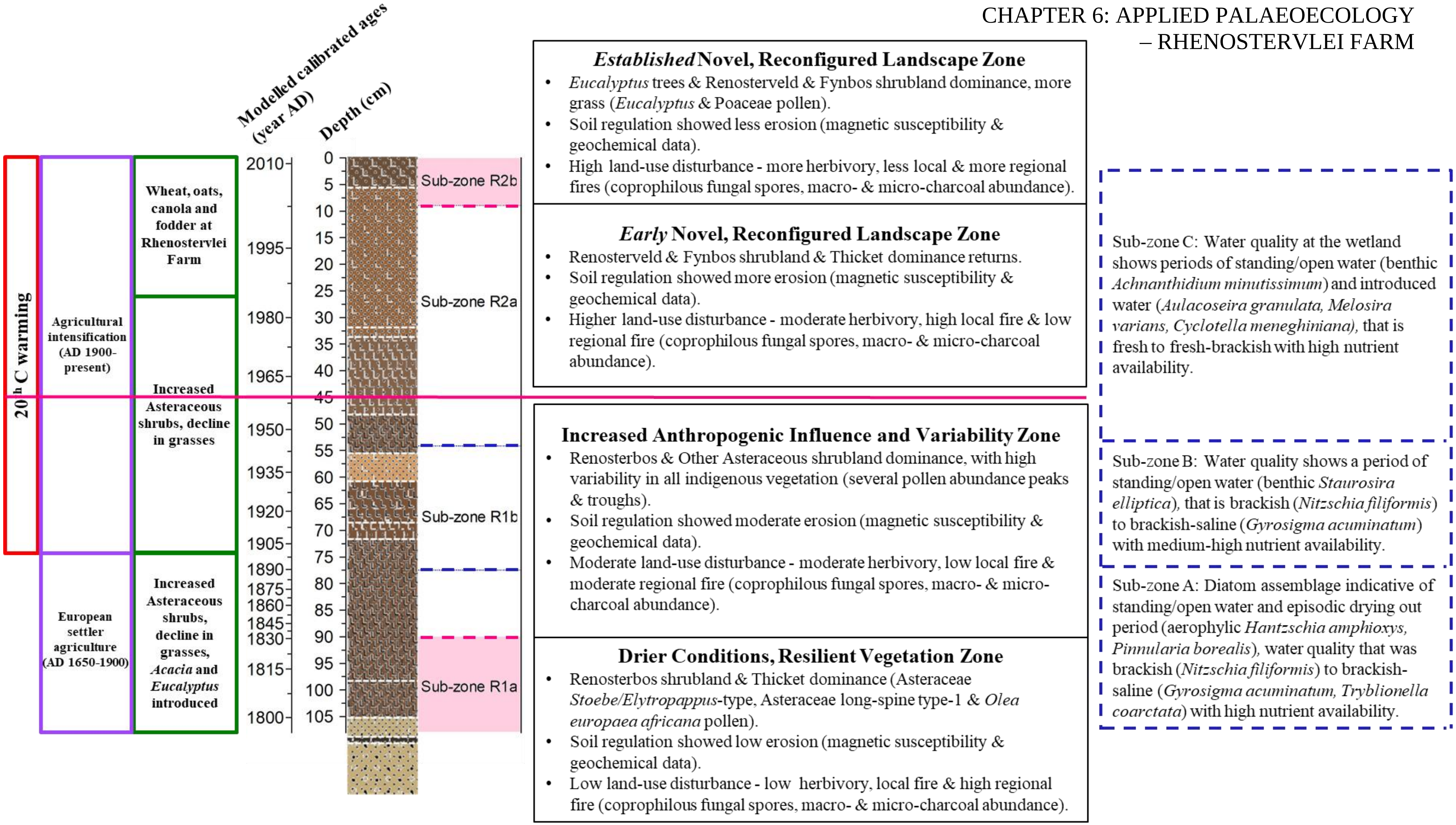



***Figure 6.13: Schematic summarising how the main findings of the RV3 core relate to long-term climatic and land-use change shown on the far left of the diagram. Pollen zonation (pink horizontal lines for Sub-zone R1a, R1b, R2a and R2b) and diatom zonation (blue dashed lines for Sub-zone A, B and C) is shown separately. Main text boxes in the middle summarise the interpretation according to the pollen zonation, whereas the supporting diatom boxes on the far right are interpreted according to the diatom zonation.***

### 6.6.1 Reconstruction and interpretation of environmental history over decadal-centennial timescales

#### *6.6.1.1 Drier conditions and minimal land-use support resilient vegetation during ca. AD 1795-1825 (Sub-zone R1a)*

The results show that plant biodiversity in ca. AD 1795-1825 (108-90 cm; Pollen Sub-zone R1a) comprised mainly of Renosterbos, Renosterveld and Fynbos, and Thicket elements. This diverse flora persisted despite evidence for dry climatic conditions shown by the presence of aerophilic taxa in the diatom assemblage in ca. AD 1795-1890, and diatoms tolerant of high salinity (108-78 cm; Diatom Sub-zone A). Low diatom preservation also suggests dry conditions. While the regional palaeo-climate record shows that the colder and wetter Little Ice Age (LIA) lasted until ca. AD 1800 (Tyson et al. 2000), regional climate data also shows increased rainfall variability since the ca. AD 1820s (Ndebele et al., 2020; Pribyl et al., 2019; Vogel, 1989) and a general warming trend over Africa since ca. AD 1880s (Nicholson et al., 2013) before the significant climate warming of the late 19$^{th}$ to early 21$^{st}$ century (Chevalier and Chase, 2015; Cronin et al., 2003; Haensler et al., 2010; Nicholson et al., 2013). The drier conditions suggested by the diatom data is consistent with the pollen data presented here by the prevalence of Oleaceae type-2 pollen which is likely wild olive (*Olea europaea subssp. africana)* pollen was different to the Oleaceae type-1 pollen. *Olea europaea subssp. africana* is a hardy, drought resistant, small evergreen tree. The modern vegetation survey (section 6.2, Appendix Figure 10.12 and Table 10.5) also showed *Olea europaea subssp. africana* and other Thicket (*Searsia* and *Kiggelaria africana*) still present today. Such bush-clump resprouting species occur in natural drainage lines in Renosterveld and favour friable soils, fire refuge habitats such as drainage lines, boulder screes and large termite mounds (Midoko-Iponga et al., 2003; Walton, 2006).

The diverse flora suggested by the pollen record suggested a heterogenous landscape resilient to drying. Indicators of anthropogenic land-use (charcoal and coprophilous fungal spores) suggest a low intensity of fire and herbivory, which may have helped buffer the vegetation assemblage against drier conditions. The minimal land-use disturbance suggested by low macro-charcoal and low coprophilous fungal spores abundance, as well as the presence of Renosterbos, Renosterveld and Fynbos, and Thicket elements, suggests no clearing of vegetation cover via burning or overgrazing that would have otherwise eroded biodiversity and possibly driven erosion during ca. AD 1795-1825.

*RV3 synthesis diagram page 1/2*

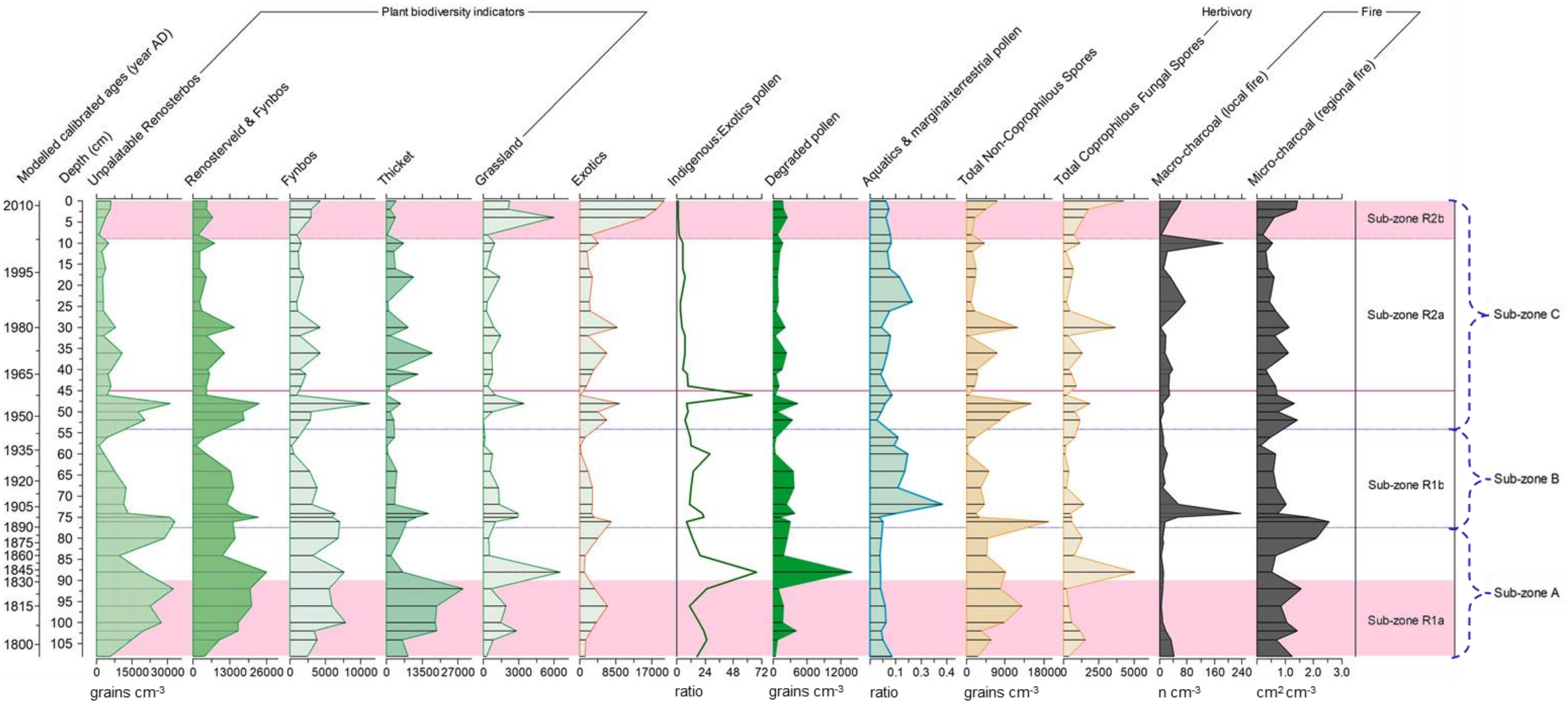
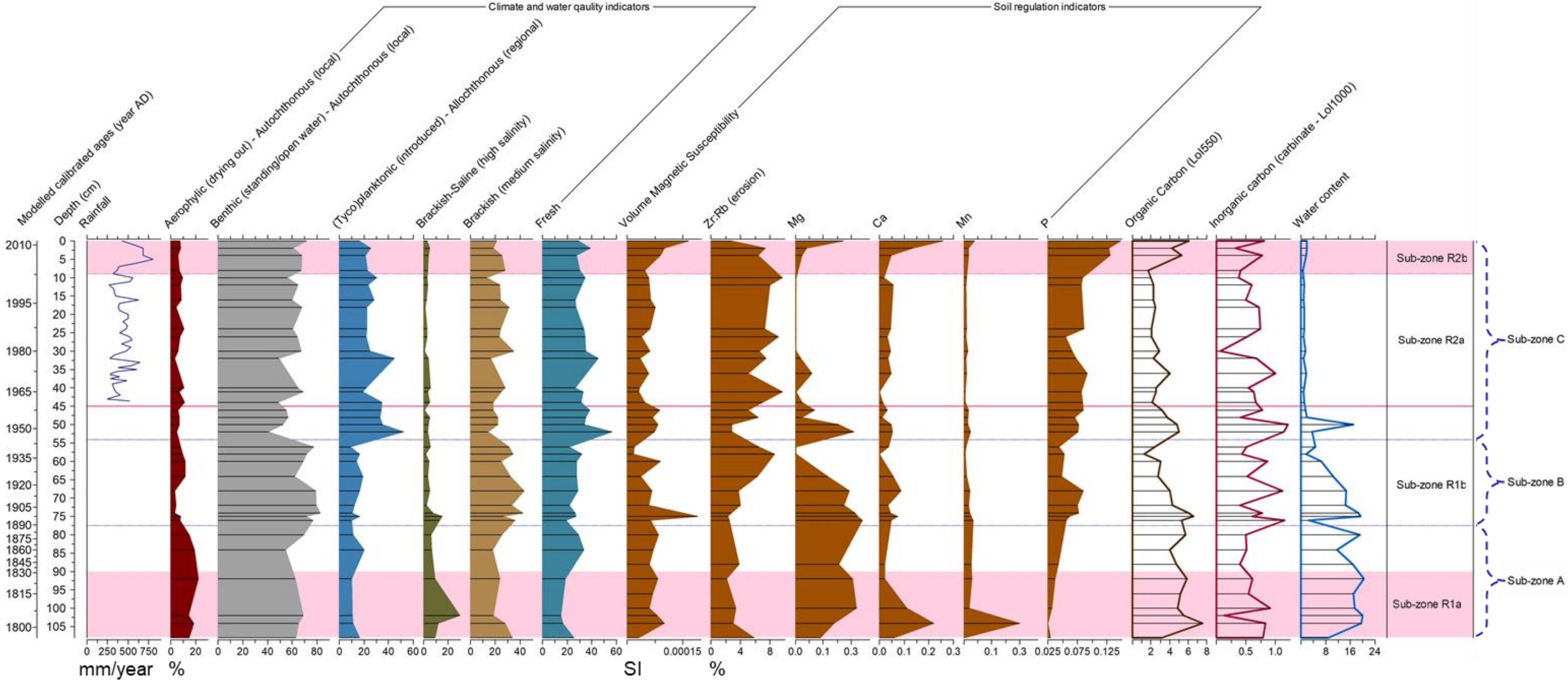



***Figure 6.14: Synthesis diagram of multiple palaeo-proxies to extend the timescale of ecosystem services (pollen and diatoms) and drivers of change (coprophilous fungal spores, charcoal, geochemical and physical properties) for the RV3 core. Pollen zonation (pink horizontal lines for Sub-zone R1a, R1b, R2a and R2b) and diatom zonation (blue dashed lines for Sub-zone A, B and C) is shown separately. The solid pink horizantal line indicates significant zonation. Pink shading indicates sub-zones (R1a and R2b) that have a similar ecological structure.***

*RV3 synthesis diagram page 2/2*



The high relative abundance of indigenous vegetation (high indigenous:exotics pollen ratio – Figure 6.13) meant that there was potentially more soil organic matter, a finding also consistent with high organic carbon ($LoI_{550}$) (Lee-Thorp et al., 2001), less bare ground (Forbes et al., 2018), and therefore less runoff water or wind-derived sediment (Van der Putten et al., 2008) from the Renosterveld shrublands and from the slopes west and south of the wetland (Figure 6.1). This inference is supported by the geochemical results, which show less erosion indicated by the lowest mean Zr:Rb ratio, clay rich sediment (Table 6.3) and low magnetic susceptibility in ca. AD 1795-1825 (Figure 6.10). Together these proxies suggest less weathering activity (Liu et al., 2004; Rydberg et al., 2016). Moreover, high aerophilic diatoms coupled with a consistent sediment accumulation rate (Figure 6.3 and Figure 6.4) suggests fewer incidents of prolonged inundation relative to other times in the record (Van der Putten et al., 2008). This is consistent with the RV3-record showing less erosion as there would not be inundation of the wetland from the adjacent Berg River during drier conditions. The low magnetic susceptibility further supports the argument for adequate soil erosion regulation at Rhenostervlei Farm during ca. AD 1795-1825.

The high resolution, multi-proxy RV3 data together support the interpretation of a thriving, drought-tolerant Renosterveld and Fynbos ecotone shrubland and Thicket patch-mosaic landscape under lower levels of land-use intensity including low local fire and grazing disturbance in the late 18$^{th}$ and early 19$^{th}$ centuries. This is indicated by aerophilic diatom taxa, Asteraceae *Stoebe/Elytropappus*-type, Asteraceae long-spine type-1 and Oleaceae type-2 pollen grouping with Zone R1's lower depths on the opposite axis of the macro-charcoal, herbivory and erosion vectors in Figure 6.12. However, the known history of farming observed in historical maps from this site and in the region (Meadows and Sugden, 1991a; River Health Programme, 2004) is dated well before the basal age of the RV3 core which was ca. AD 1795. Therefore, increased regional land-use is consistent with increased regional fires evident by the highest average micro-charcoal abundance during ca. AD 1795-1825 (Figure 6.14).

In terms of ecosystem services and resilience theory, despite increasing agricultural activities in the region and evidence of grazing and burning on Rhenostervlei Farm and dry conditions during ca. AD 1795-1825, ecological character resilience was likely preserved during this time. The pollen data suggest an intact heterogeneous patch-mosaic landscape of unpalatable Renosterbos, Renosterveld and Fynbos shrubland with Thicket elements (Oleaceae and *Searsia* spp). This high plant diversity was associated with adequate soil erosion regulation and nondetrimental water quality (medium-high salinity and high nutrient availability) (Figure 6.13 and Figure 6.14). Even though it is not possible to say from the current record what the pre-agricultural landscape and ecosystem services looked like, the evidence suggests a resilient state that was maintained by low levels of grazing and fire during drier climatic conditions.

##### *6.6.1.2 Increased anthropogenic influence and variability in ca. AD 1825-1960 (Sub-zone R1b)*

In terms of the reconstruction of environmental history and interpretation of what drove land cover change initially, decreased Renosterveld and Fynbos and Thicket pollen abundance coincided with increased herbivory as indicated by the significant peak in coprophilous fungal spores in ca. AD 1840, suggesting an increase in land-use disturbance. Therefore, only the most grazing- and trampling-resistant plants survived, as suggested by the increase in Poaceae pollen, which was previously a lesser component in the landscape (Curtis, 2013; Curtis and Bond, 2013; Forbes, 2014; Forbes et al., 2018; Newton, 2008). Increased grazing disturbance likely stimulated grass regeneration and the formation of grazing lawns at the site (Radloff et al., 2014). Evidence for increased grazing intensity is substantiated by a documented increase in stock farming in the Swartland (Table 5.1 and Table 6.1).

Though initial changes in vegetation composition seem to be associated with increased grazing pressure, unpalatable Renosterbos (Asteraceae *Stoebe/Elytropappus*-type pollen) also declined. This suggests that other factors, besides herbivory, had an influence on land cover. A drier climate was probably also a contributing factor, as suggested by the unprecedented peak in drought-tolerant Proteaceae in the record from this site, as well as a decline in drought-sensitive Ericaceae abundance (Skelton et al., 2013; West et al., 2012), and the absence of several other Fynbos pollen taxa (*Cliffortia*, *Anthospermum* and *Agathosma*). Low pollen preservation, reflected by a peak in degraded pollen, in conjunction with the aerophilic diatom abundance further supports the argument for a drier wetland environment. However, during ca. AD 1820-1880 (92-80 cm) the climate varied considerably as evidenced by increased freshwater and (tyco)planktonic diatom species, *Aulacoseira granulata (+ var. angustissima)* (Stager et al., 2012), which indicate a transition to a wetter period. Reconstructed regional paleoclimate records (Stager et al., 2012) and documentary-derived weather archives (Vogel, 1989) both support the claim of climate variability as a driver during this time seen by multiple notable wet periods in ca. AD 1844-1848, 1860, 1869-1871 1885-1900 and dry periods in AD 1824-1829, 1865, 1877-1885 and 1894-1896. Wetter conditions are confirmed by the increased abundance of drought-sensitive taxa, Ericaceae, and an unprecedented peak in aquatics and marginal:terrestrial pollen in ca. AD 1895-1905 in the record from this site . Interestingly, Juncaceae pollen, which was mostly absent in the RV3 palaeo-record, occurred for the first time in ca. AD 1900, confirming novelty in wetland conditions. These findings are consistent with palaeo-climate records that report a precipitation maximum in ca. AD 1930 (Stager et al., 2012).

The main driver of changes in water quality and vegetation is likely because of a significant fire event at Rhenostervlei Farm shown by the macro-charcoal peak in ca. AD 1900. Fire would have reduced vegetation cover (decreased pollen abundance of all vegetation types, including grass), increased runoff and therefore resulted in an open/standing water wetland environment containing more nutrients. This interpretation of landscape processes at the start of the 20$^{th}$ century is informed by unprecedented changes in geochemical results time period in the sediment sequence from this site, which show increased erosion (significant increase in MS and Zr:Rb) and nutrient-loading driving an algal bloom (Ba indicative of palaeo-productivity). A possible cause for the fire and subsequent changes in plant biodiversity, water quality and soil erosion regulation could be increased land-use disturbance. This is supported by anecdotal evidence of land-use changes at the site after the Gildenhuys family purchased Rhenostervlei Farm in AD 1906 (Table 6.1), possibly beginning with a large fire to clear the land for more intensive farming. Furthermore, increased regional fires were also evident with the highest peak in micro-charcoal abundance in ca. AD 1895. This confirms the land-use signal and is consistent with intensification of agriculture during this time (Table 6.1). This trend may have been exacerbated by an increase in regional abundance of IAPs following the introduction of Eucalyptus, Acacia and Pinus by European colonialists over the period ca. AD 1670s-1850s (Table 5.1). The introduction of these species would have contributed to the intensification of wildfires given the increased fuel load and flammability associated with these species. IAPs burn hotter with a much higher intensity compared to indigenous vegetation (Cheney et al., 2018; Kraaij et al., 2018; Van Wilgen et al., 2010).

In general, vegetation associated with mid-19$^{th}$ to mid-20$^{th}$ century (in ca. AD 1825-1960; 90-45 cm; Pollen Sub-zone 1b) was comprised of mostly Renosterbos and other Asteraceous shrubs, with fewer Thicket elements. The water quality results show increased wetting over time as indicated by a change from diatoms associate with drying in ca. AD 1825-1890 (Diatom Sub-zone A/B transition) to benthic diatom taxa associated to standing/open water in ca. AD 1890-1945 (Diatom Sub-zone B). Water quality changed again from standing or open water to fresh, introduced water indicated by increased (tyco)planktonic diatom taxa in ca. AD 1945-1960 (Figure 6.14). However, relative highs in

*Aulacoseira granulata (+ var. angustissima)* are likely indicative of deeper standing water conditions throughout the season to a few years, from greater precipitation, lower evaporation and changes in the height of the water table (Figure 6.2) (Stager et al., 2012). Wetter conditions during ca. AD 1945-1960 are possibly also due to the introduction of water from the Berg River (see section 3.1) adjacent to the wetland. Variable water levels are supported by evidence of increased pollen from marginal taxa such as *Phragmites*-type and *Polygonum*. Generally, from ca. AD 1825 and particularly since ca. AD 1945 to near present, there has been high sediment load causing turbidity, with diatom species characteristic of medium to high nutrient availability and lower salinity (Figure 6.9).

Together with changes in water quality, all vegetation groups declined initially and then increased again during the later wetter period, except for Thicket vegetation. This suggests that these woody elements are more sensitive to variability in hydrological conditions and may have a longer recovery time in a heterogenous landscape that is exposed to frequent variability. Geochemical results confirm the recovery of vegetation since ca. AD 1940. Increased vegetation cover suggests decreased erosion and runoff resulting in less nutrients and salinity in the wetland with unprecedented low abundance of B and Rb:K in the time period in the sediment sequence from this site. Furthermore, decreased macro-charcoal (lowest relative abundance in ca. AD 1950) and micro-charcoal could be caused by a general shift in fire management practices, with an effort towards fire suppression in the region associated with agricultural policy changes since AD 1930 (Table 6.1). While water availability was higher and local fire was lower, the decline in Thicket taxa coincided with an increase in exotic taxa, suggesting that the establishment of alien trees was facilitated by disturbance and reduced competition (Richardson and Bond, 1991). Variability in land-use and climate change may have allowed the shrubland flora to fluctuate and persist even though woody Thicket taxa declined.

#### *6.6.1.3 Early novel, reconfigured landscape from ca. AD 1960-2005 (Sub-zone R2a)*

A novel, patch-mosaic landscape began to emerge since the mid-20$^{th}$ century, resulting in a vegetation state comprised of Renosterveld and Fynbos shrubland with a co-dominance of Thicket taxa and increased IAP species, *Eucalyptus*, during ca. AD 1960-2005 (45-9 cm; Pollen Sub-zone R2a) (Figure 6.13). Water quality at the wetland shows periods of standing/open water (benthic *Achnanthidium minutissimum*) and introduced water (*Aulacoseira granulata*, *Melosira varians*, *Cyclotella meneghiniana*), that is fresh to fresh-brackish with high nutrient availability relative to other times. Soil regulation showed more erosion (magnetic susceptibility & geochemical data). The main drivers of change were likely caused by interacting impacts of land-use and climate variability.

In terms of land-use drivers, fire suppression facilitated by agricultural intensification on Rhenostervlei Farm before AD 1960, explains the return of Thicket vegetation that was similar to ca. AD 1795-1825 (Sub-zone R1a). The Thicket patches most likely included fire-sensitive *Olea europaea subssp. africana* as indicated by Oleaceae type-2 pollen abundance, and still found scattered in the landscape at present (section 6.2). Furthermore, agricultural intensification practices were evident in the region as Voëlvlei Dam was commissioned upstream from Rhenostervlei Farm in AD 1952 (Table 6.1). The Voëlvlei Dam wall was raised in AD 1959 and increased water abstraction and diversion from the Berg River to support agriculture and urban development likely impacted on the flooding signal at the RV3 wetland, which is adjacent to the Berg River (Figure 3.3). Thus, river dynamics influenced by land-use impacted on water quality and soil regulation and therefore plant biodiversity, however, wetting and drying of the wetland could also be related to climate variability (Figure 6.2).

In terms of climate change drivers, the latter part of the 20$^{th}$ century has been characterised by a warming trend due to anthropogenic climate change (Cronin et al., 2003; Haensler et al., 2010). The RV3 record confirms benthic conditions as freshwater inputs began to decline at the site in ca. AD 1965 as indicated by decreased (tyco)planktonic diatoms (Figure 6.14). Apart from a noteworthy peak of *Aulacoseira granulata* in ca. AD 1980, which is consistent with a flooding event at the site in AD 1977 (Gildenhuys pers. comm. 2019) (Table 6.1and Figure 6.2), the patterns in geochemical and diatom results show periodic drying. Therefore, the relic ecotonal Renosterveld-Fynbos shrubland and Thicket patch-mosaic landscape was able to persist despite initial drier conditions, also inferred by the absence of drought-sensitive Ericaceae pollen in ca. AD 1960 (Skelton et al., 2013; West et al., 2012). However, the interpretation of intensifying land-use as the main driver of plant biodiversity loss is supported by changes in soil regulation properties such as Zr:Rb, Mg, P, $LoI_{550}$ and water content (Figure 6.14). The known historical timeline of events at the site (Table 6.1), including increased agricultural intensification since AD 1984 supports the trends and patterns of environmental change indicated by multiple proxies pre-1960s and post-1960s.

In terms of ecosystem services and resilience theory, though ecologically different, the alien trees may be functionally similar to indigenous Thicket patches, in that they could contribute to basic soil erosion regulation and soil stability that maintains ecological integrity. Therefore, the Renosterveld and Fynbos shrubland and Thicket patch-mosaic persists in a more fragmented pattern as IAPs have expanded. Despite changing trends in ecosystem services, such as decreased Oleaceae type-2 pollen and increased *Eucalyptus* pollen abundance, landscape processes continue despite the new vegetation state comprising a mix of indigenous and alien trees (Hunter, 2011; Jia and Whalen, 2020).

Despite the initial peak in Thicket pollen, most-likely the drought-resistant *Olea europaea subssp. africana*, representing a recovery in indigenous trees in ca. AD 1972, pollen from exotic species also began to increase on average in since the ca. AD 1960-2005. Indicators of increased land-use suggest that a certain level of disturbance was favourable for IAPs to outcompete the indigenous vegetation assemblage in decades to follow (see section 2.1.2.2) (Milton, 2007; Wang and Liu, 2020; Weiner et al., 2019). In addition to climate variability impacts on ecosystem services, increased coprophilous fungal spores in ca. AD 1980 suggest higher grazing intensity. Patterns in plant biodiversity, water quality and soil erosion regulation remained relatively stable until the next major shock in the system as indicated by a peak in macro-charcoal in ca. AD 2005.

#### *6.6.1.4 Established novel, reconfigured landscape from ca. AD 2005-near present (Sub-zone R2b)*

Although the novel patch-mosaic landscape began to emerge since ca. AD 1960s, the reconfiguration of the new state was more apparent in the RV3 record since the early ca. AD 2000s and its establishment is apparent in the present landscape (Figure 6.1 and Figure 10.12). The vegetation comprised of Renosterveld-Fynbos ecotonal shrubland with *Eucalyptus* trees replacing indigenous trees and increased grasses. Noticeably, pollen of other Thicket taxa (*Searsia* and *Maytenus oleoides*) were absent between ca. AD 1995-2010 and since ca. AD 2005 *Eucalyptus* pollen increased exponentially. Water quality remained relatively similar to pre-2000s wetland conditions that were fresh to fresh-brackish with high nutrient availability. However, soil regulation changed indicated by increased magnetic susceptibility, decreased erosion (Zr:Rb) and unprecedentedly high P and Ca in the time period in the sediment sequence from this site.

In terms of drivers of change, the major fire event (peak in macro-charcoal) in ca. AD 2005 at the Sub-zone R2a/R2b transition marks a more established, novel landscape at Rhenostervlei Farm. Whether this fire event was a result of a planned burn or a wildfire is uncertain but it coincided with unprecedented changes in ecosystem services in the time period in the sediment sequence from this site (Figure 6.14). Increased local and regional fires could also be related to fire-vegetation dynamics exacerbated by late 20$^{th}$ century warming causing a higher FDI and increased susceptibility to wildfires (Forsyth and van Wilgen, 2008; Kraaij and Wilgen, 2014).

Although stable trends in (tyco)planktonic and freshwater diatom taxa since the ca. 1980s indicate that water quality remained unchanged in years leading up to the vegetation state transition, *Achnanthidium minutissimum* increased in ca. AD 2010 (4 cm) indicative of physical disturbance (Appendix Figure 10.19). This interpretation of increased disturbance at the site is supported by local rainfall data and anecdotal evidence as it coincides with another reported flood event in AD 2007 (Gildenhuys pers. comm. 2019) (Table 6.1 and Figure 6.2) demonstrating the compounding impacts on ecosystem services when multiple climate and land-use drivers interact. A notable peak in coprophilous fungal spores in ca. AD 2011 (0 cm) signifies prevailing land-use impacts at the site (Appendix Figure 10.12). The dramatic and unprecedented increase in *Eucalyptus* pollen during ca. AD 2005-2011 is strongly associated with herbivory, Poaceae pollen, (tyco)planktonic (regional/introduced) diatom taxa, and erosion (Figure 6.12) in the time period in the sediment sequence from this site. Furthermore, the NMDS results show that *Eucalyptus* pollen is strongly associated with increased P. Perhaps it is not surprising that there was an unprecedented increase in *Eucalyptus* since the early ca. AD 2000s as trends in soil regulation indicators including P, Mg and Zr:Rb changed gradually, yet more drastically, over centennial timescales (Figure 6.14).

In terms of ecosystem services and resilience thinking, palaeoenvironmental data characteristic of the landscape at Rhenostervlei Farm show that plant biodiversity, water quality and soil erosion regulation is impacted on by land-use and climate variability during the late-20$^{th}$ century and 21$^{st}$ century warming. Perhaps a post-fire landscape since ca AD 2005 together with benthic conditions since the ca. 1980s allowed for a conducive competitive environment that favoured *Eucalyptus* at the site. By the early 21$^{st}$ century, the system deviated from its “natural ecological character”, as typical Thicket elements disappeared, as indicated seen by IAPs dominance along the riverine fringe and scattered further inland within the Renosterveld-Fynbos shrublands (Appendix Figure 10.12). In terms of structural character of the riverine fringe, alien trees including *Eucalyptus* and *Acacia* have replaced the function of indigenous trees (*Olea europaea subssp. africana*), which were common in the late 18$^{th}$ century to early-19$^{th}$ century (ca. AD 1795-1825, Sub-zone R2b). There are however remnants of Thicket taxa still present at the coring site and at the Sandstone Fynbos ecotone (section 6.2). Therefore, even with increased variability in the changes in the drivers (climate, fire and herbivory), changes in the ecosystem services did not indicate that a critical environmental/climate threshold was exceeded within the ca. 215 years.

Implications of a novel landscape are shown by geochemical and physical properties results such as unprecedentedly high P, Ca, Zn and MS relative to other times. However, the increase in IAPs coincided with a general decline in erosion (Zr:Rb) since ca. AD 2005 (Figure 6.14). As IAPs play a major role in streamflow reductions in the Western Cape Water Supply System (WCWSS), their increased biomass would reduce water input to the wetland (Le Maitre et al., 2019, 2020, 2016). The recent decrease in Zr:Rb and increase in Mg suggests a return to pre-1830s levels of soil regulation when local land-use disturbance was lower and indigenous plant biodiversity resembled a more natural ecological character. There was also an increasing trend in micro-charcoal abundance during ca. AD 2005-2011 showing a similar abundance to that of pre-1960s regional fire levels (i.e. ca. AD 1795-1960; Sub-zones R1a and R1b).

### 6.6.2 A resilience and boundary crossing perspective at Rhenostervlei Farm

It is logical to assume that a transformed landscape largely converted to croplands and grazing lands with remnants of natural vegetation infested by IAPs will be less resilient, but what does the palaeo-evidence say about change in ecosystem services and resilience over time? Do the palaeoecological proxies indicate threshold behaviour? And what do the results mean for management?

#### *6.6.2.1 Palaeoecological insights linked to ecological thresholds and potential management thresholds at Rhenostervlei Farm*

The transition from Renosterbos-dominated Renosterveld-Fynbos ecotonal shrubland and Thicket mosaic to Renosterveld-Fynbos shrubland and Thicket mosaic with increased Eucalyptus and grass patches occurred in ca. AD 1960s. The vegetation state transition occurred as a result of both local and regional agricultural intensification with impacts exacerbated by climate variability. Statistically significant pollen zonation in the palaeo-record showed changes in the relative abundance of different vegetation components (see Figure 6.15), but all are maintained throughout the record i.e. there is no complete loss of any of the vegetation elements, and thicket elements recover after a substantial reduction during ca. AD 1825-1960. Although the vegetation state is presently outside its long-term or pre-impact ecological character, and despite a possible ecological threshold being crossed, I argue that it is not critical due to the persistence of at least 50% pollen from indigenous flora (Figure 6.15), a diverse complement of diatom taxa (section 6.5.4 and Appendix section 10.6.5) and a rich suite of typical Renosterveld and Fynbos species still present in the landscape (see vegetation survey in section 6.2 and Appendix 10.6.1). The resilience of the SES is achieved via retention of structural character over decadal-centennial timescales. This suggests that despite disturbance by fire, herbivory, drought and floods, and competition by IAPs, that heterogeneity of the landscape helps to maintain resilience (i.e. ability to recover after disturbance).

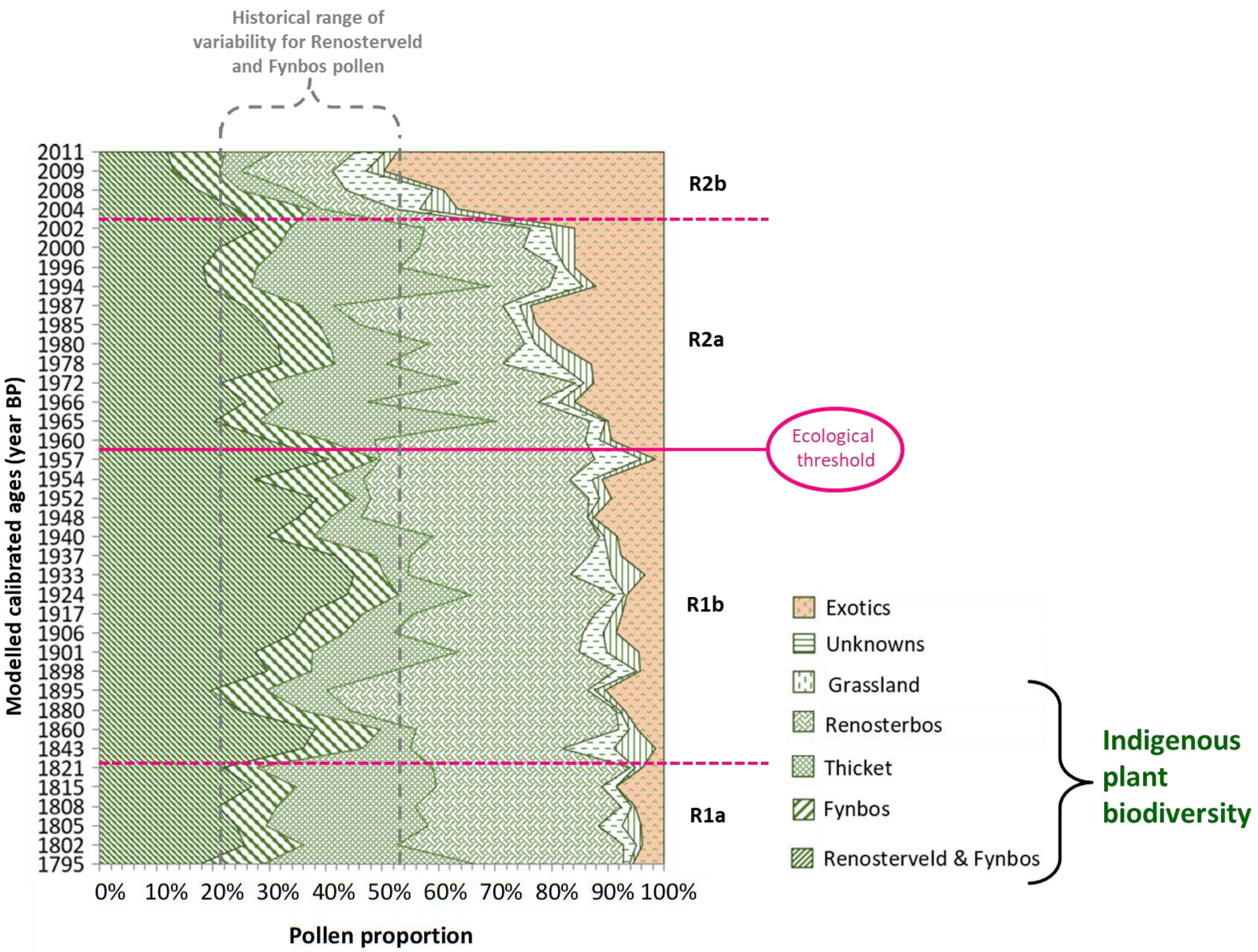



***Figure 6.15: Indigenous plant biodiversity and exotic aggregates of fossil pollen data from the RV3 core expressed as percentage values. The pink solid horizontal line separating zones R1 and R2 indicates statistically significant zonation, representative of a possible ecological threshold, while the pink dashed horizontal lines indicate sub-zones. The grey dashed vertical lines represent a historical range of variability for Renosterveld and Fynbos management based on the historical range of variability of the pollen record that could be tested in future participatory system dynamics (PSD) modelling research. The assumption is that the aggregated value of shrubland pollen should not decrease below its historically lowest value (21%) in ca. AD 2009.***

Furthermore, statistically significant pollen zonation (Zone R1 and R2 – section 6.5.1.1) is not synonymous to a regime shift to an alternate stable state since trends have not occurred in multiple ecological indicators pre- and post-1960s. An example of this is that changes in water quality were more negligible compared to the significant decrease in plant biodiversity over the ca. 215-year-old record. Despite apparent resilience of the landscape and no critical regime shift, the degree of disturbance has on average increased from ca. AD 1960s (Figure 6.13). A similar result was found at the nearby lowland conservation case study site, Elandsberg PNR, which showed a significant vegetation state transition in the mid-1950s, around a similar time to RV3 (i.e. mid-1960s). However, the VANG palaeo-record also shows a simultaneous increasing trend in fire and herbivory that suggests the development of an alternative stable state (Forbes et al., 2018).

Together multiple palaeo-proxies show that the presence of alien trees alongside altered nutrient availability and persisting shrubland and thicket elements suggests a novel and reconfigured, though still functional, landscape. It is essential that a suite of ecosystem services and drivers are analysed to obtain a richer picture of SES dynamics, particularly since multiple ecosystem services can respond differently to the same disturbance and therefore indicate delays in the system. As an example, variability in the pollen assemblage was first evident in the ca. AD 1830s (Pollen Sub-zone R1a). However, variability in the diatom assemblage began in the ca. AD 1890s (Diatom Sub-zone A/B transition) and ca. AD 1940s (Diatom Sub-zone B/C). This variability was prior to the statistically significant shift in pollen data in the ca. AD 1960s (Pollen Zone R1/R2 transition). The differences in the timing of RV3 pollen and diatom zonation show potential detection of system delays and highlights the sensitivities of this SES, where significant changes in plant biodiversity have taken place but not significant changes in water quality. The VANG palaeo-record at Elandsberg PNR shows similar system delays (Forbes, 2014; Forbes et al., 2018). Noticeable increasing trends in coprophilous fungal spores began in the early ca. AD 1800s during the European settler agriculture period, and persisted through the agricultural intensification period (ca. AD 1900-1970s) and conservation period (1970s-present). Despite prolonged grazing disturbance, significant pollen zonation only occurred in ca. AD 1950s as seen by the dramatic increase in Asteraceae *Stoebe/Elytropappus*-type (unpalatable Renosterbos), coupled with a change in fire regime (unprecedented increased macro-charcoal). Furthermore, the pollen zonation results show a decrease in the range/duration of change - i.e. from slower to faster rates of change in ecological groupings. In this case, ca. 130 years pass between the R1a/R1b to R1b/R2a transition, whereas only ca. 45 years pass between the R1b/R2a to R2a/R2b transition. With such results, the question arises as to whether resilience is compromised as the range of system delays decreases (see section 6.6.2.2 below)?

Plant biodiversity was resilient to hydrological fluctuations, until intensification of human disturbance associated with more local fires, herbivory and land-clearing caused by agricultural intensification. Intensification of these drivers, together with increasing presence of non-indigenous species (IAPs such as *Eucalyptus* and *Acacia*), with feedbacks on soil regulation and ecological function, resulted in an established novel, reconfigured ecosystem in the early 21$^{st}$ century (Pollen Sub-zone R2b). The persistence of the heterogeneous patch-mosaic landscape and continued landscape processes via niche conservatism of the new vegetation state creates functional redundancy, and the maintenance of multiple ecosystem services (supporting/provisioning and regulating) (Gillson et al., 2020; Hunter, 2011; Jia and Whalen, 2020; Oliver et al., 2015; Yachi and Loreau, 1999) that continues to support agriculture and buffers the SES from environmental change. Functional redundancy is important in terms of land-use management within the highly fragmented lowlands and it is essential to consider what the historical range of variability is and the limits of acceptable change for conservation agriculture and choosing appropriate reference conditions (e.g. pre-1830s or pre-1960s) for setting realistic restoration and river rehabilitation targets.

#### *6.6.2.2 Further research and implications for management decision-support at Rhenostervlei Farm*

Since the commission of the Voëlvlei Dam in AD 1952, changes in water supply whether due to climate variability and dam construction both upstream and downstream affects ecological processes at the wetland where RV3 core was retrieved (section 3.2.2.2). Therefore, it too could have an influence on water release rates, the water table position and the amount of standing/open water, especially as Rhenostervlei Farm is downstream from Voëlvlei Dam (see section 6.6.1.3 above). Furthermore, the spread of IAPs puts further pressure on water supply and demand in the Berg River Catchment, of which 5% is already utilised by alien vegetation (Cheney et al., 2018; Midgley et al., 2014; Pegram and Baleta, 2014; van Wilgen et al., 2008) (section 3.1.3). If continued land-use and IAPs infestation continues at the rate it has since ca. AD 2000s then a transition to a degraded alternate stable state might be a possibility in future (Forbes et al., 2018; Slingsby et al., 2014). Increased pressure from anthropogenic-induced climate change including further declines in rainfall and extreme weather events (Le Maitre et al., 2019, 2020) can alter this critical ecological threshold with additional climate change uncertainty (Hoffman et al., 2011) making the timing of change even more unpredictable. Recent climatic events such as the 2015-2018 regional-scale drought of the Western Cape impacted on the City of Cape Town's municipal apply including agricultural operations (LaVanchy et al., 2019; Pascale et al., 2020), and thus provides an example of how a lack of climate-resilient agriculture and alternative livelihoods can affect SES resilience. The landowner of Rhenostervlei Farm (see CF:1 in Appendix Table 10.1) expressed their concern about the climate change impacts they experienced, "*During the 2018 summer drought they [government via the Western Cape Water Supply System] had to cut our water supply by 60% and we could only fill 5 out of 15 waterblommetjie dams*." (Gildenhuys pers. comm. 2020). Although waterblommetjies are grown as a crop in several small dams on Rhenostervlei Farm (section 3.2.2.2), extreme climatic events such as the recent drought impacted significantly on their irrigation needs to keep production going. Furthermore, warmer water temperatures during warmer winter months (Allsopp et al., 2014b; Hoffman et al., 2011)were reported by the landowner to be inadequate to induce waterblommetjies' flowering (Gildenhuys pers. comm. 2020). This is detrimental to the harvest as the inflorescences is the part of the plant that is sold for consumption.

The RV3 multi-decadal temporal perspectives provides valuable insights for further examination of trends and rates of change in multiple system variables over relatively extended timescales, and cross-comparison between rates of different ecosystem variables such as plant biodiversity and water quality. An example is the acceleration in rates of long-term vegetation change (Table 6.4) as well as the difference in pollen and diatom zonation results (Table 6.4, Table 6.5 and Figure 6.14). In systems thinking and SD terminology, these 'system delays' potentially lead to the emergence of side effects in future, which can be explored through the identification of feedback loops, delays, and non-linearity by the explicit representation of stocks and flows (section 2.4.2). In addition, the RV3 record supports the assumption that a shift from natural Fynbos vegetation to Grassland is a disturbance indicator (Neumann et al., 2011). Poaceae pollen abundance was the highest at the beginning of the transitional period in ca. AD 1840, together with increased grazing, perhaps an indication of factors altering the fire regime and the sensitivity of this SES. Interestingly, the proposed 130-year transitional period (ca. AD 1825-1960) was also marked by two high peaks in the indigenous:exotics pollen (Figure 6.14). Since ca. AD 2000s there is evidence of modulation and potential divergence of the abiotic environment such as *Eucalyptus* creating a tall forest, grassland facilitating fire and changes in soil regulation. The obvious change in ecological character at Rhenostervlei Farm has resulted in the riverine fringe being dominated by *Eucalyptus* at present (Figure 6.1, Appendix Figure 10.12 and Table 10.5).

The historical range of variability for Renosterveld and Fynbos vegetation pollen in the RV3 record at this site is 21-53% historical range of variability between ca. AD 1795-2011 (Figure 6.15). If the percentage pollen data is calibrated with quantitative estimates of current indigenous vegetation cover at Rhenostervlei Farm, then it too can inform the setting of management thresholds in collaboration with the landowner and conservation practitioners who would decide what limits are acceptable in the landscape. The pollen and geochemistry data indicate levels of *Eucalyptus* pollen and P much higher since ca. AD 1960 than they had been in the previous ca. 165 years. Thus, a possible management threshold might be to return soil erosion regulation to pre-1960s levels, a time when Renosterveld and Fynbos pollen was relatively higher and erosion relatively lower in the sedimentary sequence at this site. If the variability falls below the historical range of variability, then it could be more likely that a critical degradation threshold could be crossed (Figure 6.15). The loss of plant biodiversity could result in IAPs becoming biotic modifiers that impact the ecosystem by altering the structure and function of the biotic and abiotic environment (Le Maitre et al., 2011; Linder et al., 2012), resulting in an alien-dominated alternative stable state. However, land-use managers would need to consider a palaeo-safe operating space for the role Thicket elements (*Oleaceae* and *Searsia*) play in to river fringe stabilization and water quality (see quotes from stakeholders in Figure 6.16), or the impacts of undesirable Renosterbos homogenization on SES resilience in the landscape (Table 10.5) (Forbes et al., 2018).

Long trends in processes allow comparison of directions and relative rates of change that may help identify critical points of inflection (Dearing et al., 2012; Steffen et al., 2006), therefore help us define the historical range of variability and recognise critical thresholds for resilience. Given the relatively short timescale of the RV3-record, which only begins after European settlement agriculture, our study does not exclude the possibility of a pre-colonial (pre-1600s) record providing additional information on fast and slow ecological processes that may alter our perception of the resilience at Rhenostervlei Farm. A longer record and further rigorous quantitative analyses of the multi-proxy RV3 palaeoecological data could be conducted in future to test for a possible regime shift in the past (Seddon et al., 2014) and/or a potential regime shift occurring in the not-too-distant future. These insights may be an indication that current SES function and resilience at Rhenostervlei Farm and the surrounds is not sustainable in the medium- to long-term future. This finding is consistent with research that shows long-term and delayed impacts of IAPs on Fynbos plant diversity, with existing evidence for the genitive effects of altered biodiversity composition on ecosystem functions and ecosystem services (Slingsby et al., 2017, 2014).

Future management needs to maintain heterogeneity and ensure that similar sites in the region have functional connectivity for effective biodiversity conservation (Hodgson et al., 2009; Jia and Whalen, 2020). Plant biodiversity, a key provisioning and supporting ecosystem services in the CFR, is now outside of its historical range of variability (i.e. prior to the 1960s) in terms of declining Renosterveld, Fynbos and Thicket elements (Figure 6.15). This suggests a degradation of ecological character (Gell, 2012; Gell et al., 2013) over the past ca. 215 years at Rhenostervlei Farm, and we may perceive that ecosystem function and resilience is diminished. Nevertheless, as multiple vegetation elements are retained, suggesting that as yet no critical ecological thresholds have been crossed, this would need to be validated with a longer record that extended to the time before European settlement and modern pollen calibration. The diatom data suggests that water quality has remained resilient over time despite disturbance variability since the late-18th century. The landscape appears to be disturbance adapted, with a possible analogous role of domestic herbivores (Figure 6.14 and Figure 10.12) replacing large indigenous herbivore (LIH) guilds, therefore remaining ecologically functional, though a longer record would be needed to establish the nature of ecosystem services prior to colonial impact to determine whether the current landscape is still within a safe operating space.

Understanding SES resilience requires integration of environmental, ecological and social aspects of ecosystem change, and using mixed methods to harness multiple knowledge streams is necessary. Furthermore, temporal and spatial variability requires an adaptive approach to sustainable land-use management in the CFR. With the systems thinking definition and purpose of mental models in mind, capturing multi-stakeholder perspectives and narratives of change provides a contextual basis for gaining insights to possible leverage points within systemic structures (see lowest levels of the Iceberg Model in Figure 2.3) (Monat and Gannon, 2015). Land-use managers shared their perceptions of change over time and the interacting social-ecological connections (Appendix Figure 10.24) during semi-structured interviews and the multi-stakeholder engagement workshop. Several noteworthy quotes of perceptions on water quality as an ecosystem service are shown in Figure 6.16, which are particularly insightful for management of the lowland agricultural site. It highlights potential problem definitions related to how water quality impacted on by IAPs, climate change causing more evaporation and salinity, nutrient load from dung due to overgrazing, and the connections between fire and water. This serves as an example of the local contextual knowledge needed to augment the evidence-based understanding of SES resilience derived from the long-term palaeo-respective and apply multiple perspectives in adaptive management.

Although the Berg River Catchment is a widely studied area, with several plans and programmes contributing to the comprehensive understanding of its ecological systems and processes (section 3.4.2), applied palaeoecological evidence is lacking (Figure 2.2). To the best of my knowledge, RV3 is the only high temporal resolution multi-proxy palaeo-record besides the VANG core from Elandsberg PNR (Forbes et al. 2018). The results suggest that land-use in terms of grazing and burning might be compatible with sustainable ecosystem services provision (Table 1.2) and it is recommended that land-use managers plan for a pre-1960s restoration target to maintain current and future ecological integrity for sustainable water and soil regulation, two essential regulating ecosystem services that support productivity at lowland agricultural sites such as this. In addition to restoring the small fragments of indigenous vegetation at Rhenostervlei Farm, a systemic effort is needed to improve connectivity and management within the agroecosystem at a larger scale (see below). However, a caveat of this recommendation is that a longer record is also examined in order to confidently establish management thresholds. Two major peaks in macro-charcoal occurred during the RV3 record, whether these were planned burns or wildfires cannot be verified. However, the occurrence of wildfires are not uncommon in the region. The land-use manager mentioned that wildfire had occurred in recent years post retrieval of the RV3 sediment core in AD 2011 (Gildenhuys pers. comm. 2019). This anecdotal account was verified by the estimated age of the community mostly *Leucadendron* spp and *Leucospermum rodolentum* found in vegetation unit (VU 5) during the modern vegetation survey (Table 6.2 and Figure 6.1). Rhenostervlei Farm that contains vegetation remnants with irreplaceable plant conservation value as they contain rare and endangered proteas and indigenous Thicket elements.

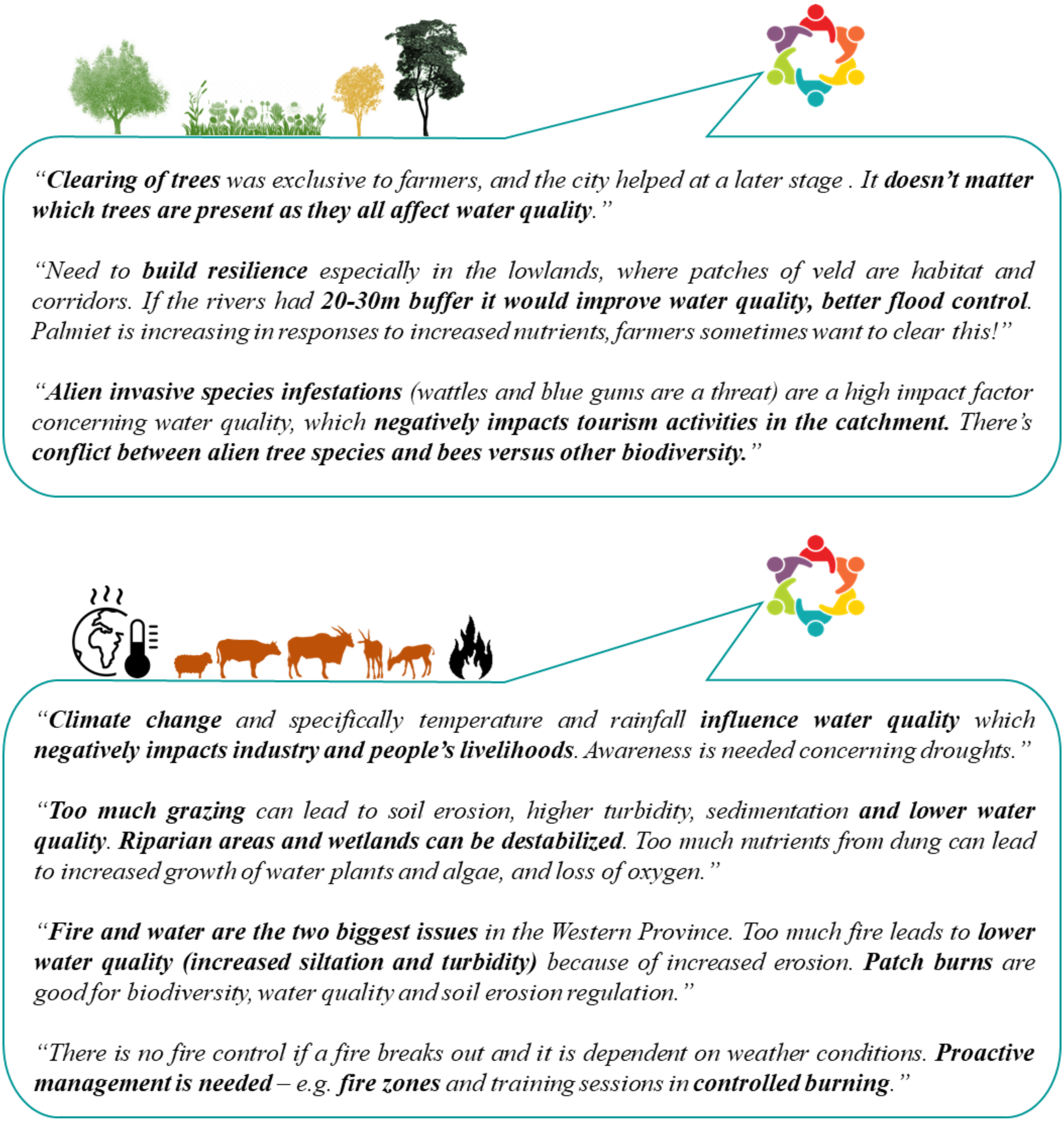



***Figure 6.16: Stakeholders' perceptions on water quality within the Middle Berg River Catchment of the CFR.***

Ecological restoration, defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" plays a role in connecting social, community, productivity, and sustainability goals (Gann et al., 2019). Therefore, ecosystem-based management and restoration recommendations for Rhenostervlei Farm include: (1) Restoring fragments of indigenous vegetation in agroecosystems, (2) Clearing of alien species and restoring thicket stands in the riverine fringe and floodplain. Thicket taxa could include e.g. *Olea europaea subssp. africana, Searsia, Dodonaea, Maytenus*, Afromontane taxa such as *Heeria argentea* or fynbos-forest ecotonal vegetation such as *Psoralea, Kiggelaria* and *Virgilia* species along the riverbank and floodplain of the Berg River and its tributaries. (2) Remnants of the natural vegetation can be found on soils that are too poor for cultivation or steep slopes that are unsuitable for planting. However, even these remnants are used for grazing of cattle. Therefore, less intensive grazing with a well-monitored rotational grazing regime that protects indigenous plant biodiversity is ideal. (3) Improved fire control (not complete fire prevention). (4) In line with conservation stewardship, the restoration of inactive croplands close to the river to aid systemic Berg River rehabilitation efforts. (5) Monitoring of water quality (characterised by diatom species affiliated with medium to high nutrient availability and low salinity) and soil regulation indicators (e.g. lower Zr:Rb, P, Zn and Ca, and higher Mg and Fe) is also recommended to manage resilience to future climate and land-use change.

### 6.6.3 Concluding remarks for Rhenostervlei Farm SES

This study highlights the importance of Berg River dynamics for the interpretation of palaeoecological findings for informing biodiversity conservation, land and water management to achieve an integrated catchment management approach in both policy design and implementation.

Evidence of degraded plant biodiversity, reflected at the site as a significant shift from indigenous to alien vegetation since ca. AD 1960, may be indicative of ecological degradation from a traditional biodiversity conservation perspective. However, the vegetation state change at Rhenostervlei Farm is representative of structural reorganisation and functional traits buffering impacts by land-use intensification. Despite the high variability in fire and grazing disturbance coupled with the variability in climate over time, plant biodiversity and water quality, and ecosystem resilience and function has been maintained. Since the early ca. AD 2000s novel ecological parameters were evident: increased alien trees and grass, herbivory, a moderate local fire regime, high magnetic susceptibility, Mg, P and Ca and recently decreased erosion. Based on the multi-proxy palaeoecological evidence, a possible restoration target could include the clearing of IAPs and their replacement with native thicket vegetation to safeguard remnants that irreplaceable plant conservation value in this transformed landscape. I propose that future research include the development of a SD simulation model for Rhenostervlei Farm by exploring feedback loops, system delays and non-linearity and uniquely testing future scenarios. Research outputs can be used in future engagement with multi-stakeholders to gain further understanding of complexity and non-linear SES dynamics within the CFR.

# 7 CHAPTER 7: COMBINING PALAEOCECOLOGY AND PARTICIPATORY SYSTEM DYNAMICS AT ELANDSBERG PNR

This chapter addresses Objective 4, Components B and C (Figure 1.2 and Table 1.1) and provides a proof of concept for applying a past-present-future lens of environmental change to inform conservation of a biodiversity-rich, but endangered, ecosystem at Elandsberg Private Nature Reserve (Elandsberg PNR). While there is merit palaeoecological studies over timescales of centuries to millennia, this case study emphasises the importance of using multi-proxy palaeoecological findings to advance the process-based and prescriptive nature of resilience theory. This is done by blending applied palaeoecology and participatory system dynamics (PSD) modelling methods for informing sustainable land-use management (Process Steps 5-12, Table 4.5). Palaeoecological data from the ca. 1300-year-old VANG sediment record was used to define the historical range of variability (Forbes, 2014; Forbes et al., 2018). PSD was used to articulate dynamic feedbacks and conduct an analysis of future scenarios. Here we present a model of ecosystem change in response to grazing and fire using palaeoecological data as a reference mode over the past 1300 years. Together, palaeoecological and PSD insights are useful for articulating both an evidence- and process-based understanding of resilience. Does the model generate any simulation results that are surprising and how do we use this process-based knowledge to inform land-use management and resilience in the region?

## 7.1 Introduction: Relevance of palaeoecological and system dynamics (SD) research in conservation landscapes

The aim of this component of the study is to demonstrate the application of the proposed conceptual meta-framework, past-present-future lens of environmental change (Figure 1.2). The concept of resilience is closely linked to sustainability, because exceeding resilience thresholds can lead to degradation and loss of ecosystem services (Carpenter et al., 2001; Chapin et al., 2010; Lew et al., 2016; Marchese et al., 2018; Quinlan et al., 2016). Therefore, resilience thinking has potentially a significant contribution to make to sustainability-oriented impact assessments (Bond et al., 2015). However, it is important to note that resilience is not always a desirable characteristic in itself since several unsustainable and undesirable states are highly resilient due to their resistance to change or reversibility (e.g. “negative resilience” exemplified by social constructs such as poverty, crime and corruption or degraded ecological regimes such as invasive alien infestation and lake eutrophication) (González Sagrario et al., 2020; Lake, 2013; K. G. Turner et al., 2016).

Using techniques and tools to understand the hidden complexity of SES can provide a sense of how resilient a system is. Resilience is core to system dynamics (SD) because it emphasizes non-linear processes, thresholds, uncertainty and surprise, as well as how periods of gradual change interplay with periods of rapid change across temporal and spatial scales (Folke, 2006). Past studies demonstrate the successful application of system dynamics in environmental management interventions (Beall et al., 2011; Brent et al., 2017a, 2017b; Carnohan et al., 2021; Clifford-Holmes et al., 2017b, 2017a; Menendez et al., 2020; Stave, 2003; Turner and Kodali, 2020; Winz et al., 2009). Furthermore, recent system dynamics research in food systems has shown its importance in influencing the resilience and overall sustainability narrative (Jose and Kopainsky, 2020; Meuwissen et al., 2019; Stave and Kopainsky, 2015).

Resilience theory topics that can be explored using palaeoecology include transitions to alternative stable states (Holling, 1973; Lebel et al., 2006), historical range of variability (Jeffers et al., 2015; Wolfe et al., 2007), and Threshold of Potential Concern (TPC) or Limits of Acceptable Change (LAC) (Gillson and Duffin, 2007; Newall et al., 2015; Rogers and Biggs, 1999). Palaeoecological studies have demonstrated the importance of long-term data for informing land and water management and biodiversity conservation (Birks, 2012; Dearing, 2008; Dearing et al., 2012; Gell, 2012; Gell et al., 2005; Gil-Romera et al., 2010; Gillson and Willis, 2004; Jeffers et al., 2015; Pederson et al., 2006; Willis and Birks, 2006). A long-term palaeo-perspective provides improved evidence of historical variability, a unique opportunity to identify drivers of environmental change, resilience thresholds and can inform potential management thresholds when palaeo-data is calibrated with land cover parameters and stakeholder’s set TPCs in their strategic management plans.

Research that combines long-term palaeoecological data with system dynamics is rare, with the exception of the agro-ecosystem study conducted in the United Kingdom (Armstrong McKay et al., 2019). Therefore, the present study’s methodological approach to extend the temporal scale and use multiple-proxy palaeo-data as reference modes during model development is unconventional in the SD field and SES research, and is the first of its kind within the CFR, SA and the African continent. This novel methodological approach (palaeoecology, stakeholder engagement and system dynamics) has the potential to provide management decision-support for land-use managers as it considers the historical range of variability and therefore a possible “palaeo-safe operating space”. Though quantitative reconstruction of ecosystem services and drivers is still in development stages and the palaeo-data (pollen, coprophilous fungal spores and macro-charcoal) needs to be calibrated, comparing recent (i.e. post-1950s) ranges of historical variability with those at a longer timescale (ca. AD 750-1950) is useful in testing potential management thresholds (e.g. Figure 5.11 and Figure 6.15Figure 6.15) using a simulation experiment.

### 7.1.1 A proof of concept that applies a past-present-future lens of environmental change

Using applied palaeoecological techniques is a useful source of long-term data when reflecting on the principle-based approach for evaluating and operationalising resilience goals (Helfgott, 2018; Hermans and Knippenberg, 2006; Herrera, 2016; Lebel et al., 2006; Pope et al., 2017). Thus, gaining an evidence-based and descriptive understanding of ecosystem resilience is beneficial for setting ecologically-realistic management thresholds for sustainable land-use within the CFR. Although blending these mixed methods is still under development, I place my study within the wider sustainability policy context through the concept of ecosystem services and therefore the necessity for mainstreaming applied palaeoecology into biodiversity conservation and other non-environmental sectors such as agriculture and urban planning in integrated catchment management. While embracing a general overarching systems-thinking conceptual framing that recognises the interconnected and dynamic nature of complexity-based SES (section 2.1), I used PSD to explore the feedback loops and further interrogate the palaeoecological data to understand the process-based and temporal complexity of changes in ecosystem services and drivers. Specifically, the approach considers temporal variability of plant biodiversity in response to environmental (climate change and fire), biotic (grazing and unpalatable Renosterbos homogenisation) and social drivers (land-use policy and fire management) and the closed loop feedbacks that have caused system vulnerability over a decadal-millennial timescale.

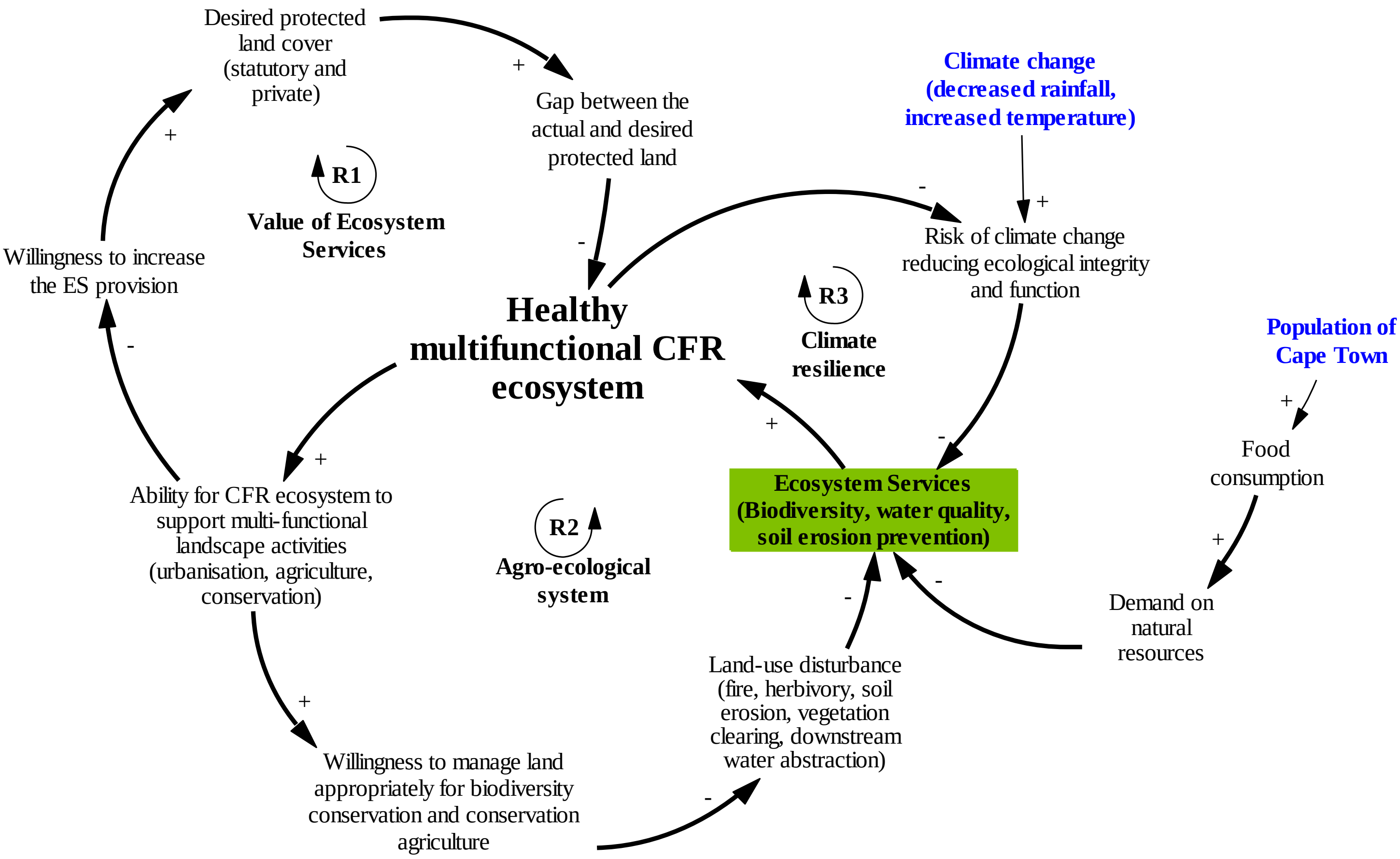



***Figure 7.1: Aggregate causal loop diagram depicting the dynamic hypothesis for a healthy multi-functional CFR ecosystem. Arrows connect two or more variables of interest and are causal links that run in the stated direction, '+' or '-'. '+' is a positive relationship, indicating that the causality runs in the same direction (i.e., an increase in variable X will cause an increase in variable Y or a decrease in variable X will cause a decrease in variable Y); '-' is a negative relationship, indicating that the causality runs in the opposite direction (i.e., an increase in variable X will cause a decrease in variable Y or a decrease in variable X will cause an increase in variable Y).***

The aggregate causal loop diagram (CLD) shown in Figure 7.1 was constructed using causality and polarity to describe feedback loops and endogeneity and it served as an initial CLD to conceptualise the intention of the modelling component of this study, depicting a high-level representation of variables and feedback loops while focusing on the bigger picture (Pruyt, 2013). It was not co-constructed with stakeholders, but it was informed by literature (see section 4.2.3.1). It maps out the broader concept of ecosystem services and ecological integrity and function in agro-ecological systems and helped to orientate the SES problems related to unsustainable land-use when climate and land-use disturbance degrades ecosystem services (section 2.4.1) (Stave and Kopainsky, 2015). The causal relationships between climate, land-use disturbance and management of ecosystem services underpin the systemic structure of the SES problems that make the goal of transitioning to a healthier, sustainable multi-functional landscape difficult to achieve or maintain. Although feedback loops act simultaneously it is helpful to consider them one at a time. The polarity of the arrows indicates the effect of one variable on another. For example, in the reinforcing loop of the agro-ecological system (R2): An increase in the ability for the CFR to support urbanization, agriculture and conservation leads to greater willingness to manage land appropriately (all other things being constant) so the causation is positive. An increase in land-use disturbance within agro-ecological SES leads to lower levels of ecosystem services provision (all other things being constant) so the causation is negative.

Furthermore, the aggerate CLD with high-level variables depicted in Figure 7.1 was constructed using causality and polarity to indicate feedback loops and endogeneity and based on literature that articulates sustainable land management (Bennich et al., 2018; Brzezina et al., 2017, 2016; Herrera de Leon and Kopainsky, 2019; Menendez et al., 2020; Stave and Kopainsky, 2015; B. L. Turner et al., 2016; Turner and Kodali, 2020), and is elaborated upon in the detailed systems map (Appendix Figure 10.24) that was co-developed with stakeholders. Therefore, in addition to the detailed systems map was used to verify the links between structure and behaviour of the SES and land-use governance shown in the aggregate CLD in Figure 7.1. For example, the variable ‘climate change’ in the aggregate CLD, includes ‘increased temperature and decreased rainfall’, while climate change predictions for the region where the three study sites are located also consider the decline in pan evaporation and wind run and extreme weather events such as floods and droughts events (Haensler et al., 2011; Hoffman et al., 2011; Meadows, 2006; Turpie et al., 2002) (section 3.1.2). Furthermore, factors such as increased funding, skills, novel methods, data analysis and translation for practical use, farmers’ buy-in and compliance with land legislation were identified by stakeholders (see Appendix Figure 10.24) and are represented here as a higher-level representation of the ‘willingness’ to manage land appropriately and value ecosystem services. Moreover, the palaeo-component is represented by the variable ‘ecosystem services’ as examples of the three selected palaeo-proxies for ecosystem services used in the present study (Table 4.4): fossil pollen, diatoms, and sediment properties and geochemical markers to reconstruct changes in plant biodiversity, water quality and soil erosion regulation (Dearing et al., 2012; Jeffers et al., 2015).

### 7.1.2 Key stakeholders and problem statement of the Elandsberg PNR case study

With implementation or management in mind, more researchers are designing inter- and transdisciplinary projects that have relevant and applicable outputs that can be integrated into decision-making and governance while considering systemic and lasting change (Helfgott, 2018; Jose and Kopainsky, 2020; Meuwissen et al., 2019; Roux et al., 2017; Sellberg et al., 2021). Elandsberg PNR, a lowland conservation site in the Middle Berg River Catchment of the CFR (Figure 3.4), is a mid-elevation site with intermediate disturbance that provides biodiversity and associated nature tourism. By using it as a case study and applying the full 13-step process (Table 4.5) , we show proof of concept for how PSD can be used to translate long-term data on ecosystem services and drivers into a usable format (e.g. decision-support tool as a boundary object for landscape and ecosystem managers) (Jeffers et al., 2015). A boundary object is information, such as specimens, artefacts, maps, models., which can be used to facilitate dialogue, engagement and learning amongst multiple stakeholders (Fischer and Riechers, 2019; Star and Griesemer, 1989).

#### *7.1.2.1 Stakeholders' perceptions*

Since Elandsberg PNR contains one of the largest areas (ca. 1000 ha) of intact West-Coast Renosterveld (Figure 3.5), it is an important site for biodiversity conservation. West-Coast Renosterveld is the most highly threatened vegetation type in the CFR since 91-97% of Renosterveld has been transformed mainly due to agriculture (crop cultivation and inappropriate grazing and fire regimes) (Rouget et al., 2003a; von Hase et al., 2003). Only 2% of West-Coast Renosterveld is formally protected and remaining fragments are isolated, critically endangered and under significant pressure because of habitat transformation. This proof of concept is of relevance to all land-use management stakeholders that participated in the stakeholder engagement component (multi-stakeholder social-learning workshop and key informal interviews). However, the target stakeholders include reserve managers at Elandsberg PNR. The simulation model is based on how they may perceive the conservation problem in terms of land-use management, including biodiversity conservation and ecotourism.

Given the stakeholders' and co-facilitators' (termed as Table Hosts) low level of experience in PSD, an established Group Model Building (GMB) script from Scriptapedia was adapted for use in this diverse community context (https://en.wikibooks.org/wiki/Scriptapedia/Variable_Elicitation)

(Wilkerson et al., 2020). The adapted script facilitates consensus-based group discussion about the environmental problems, namely, how ecosystem services are influenced and the connectedness in the system, and how the SES problems are related to sustainable land-use management within the Middle Berg River Catchment. The multi-stakeholder engagement workshop was designed to include the general principles used in World Café (http://www.theworldcafe.com), Sustainability Dialogues (Carson, 2011; Currie, 2018) and a social learning process (Reed et al., 2010). This helped different stakeholders to articulate their personal and organisational mental models related to the SES problems. Participants explored questions, encouraged and valued each other's contribution, connected diverse perspectives, and listened together for patterns, insights and deeper questions, sharing collective insights that may be actionable in future (Carson, 2011). Given the design of the multi-stakeholder engagement workshop (see Dialogues 1-3 in Appendix Table 10.2), 10-15 participants were ideal (excluding the facilitation team – lead facilitator/modeller, note taker and three Table Hosts).

Invited stakeholders were identified and selected based on their roles as land-use managers and representation from the following sectors: commercial farming (four affiliations), conservation practitioners (four affiliations) and local, regional and national land-use governance authorities (three affiliations) (Clifford-Holmes, 2015; Kumar et al., 1993; Voss et al., 2002). The location of the workshop, which was halfway between Cape Town and Porterville in the Western Cape, was to strategically increase the ability of invitees to participate. However, only seven stakeholders attended despite 13 confirmations prior to the workshop. As a part of the ethical consideration in the study (see Appendix Figure 10.4), research participants were informed about their voluntary participation, the potential risks and benefits of their participation, and the freedom for them to refuse to participate or withdraw from the study at any time without prejudice (Appendix Figure 10.4).

Despite farmers being identified as primary stakeholders in the study given that changes in ecosystem services and drivers of change would affect the agroecosystems they manage, the low turnout at the workshop could be related to dimensions around 'personal interest. Moreover, this study lacked some considerations needed for the design and implementation of inter- and transdisciplinary research that could be addressed in follow-up research. For example, the co-design of the research approach, and sufficient time, funding and flexibility to adequately establish enduring partnerships with stakeholders (Shackleton et al., 2022). However, to compensate for the low representivity of commercial farmers at the multi-stakeholder engagement workshop, methodological adaptations and flexibility was required to addressing such social issues when soliciting stakeholder involvement (Carnohan et al., 2021; Clifford-Holmes et al., 2016; van den Belt, 2004; Voinov and Bousquet, 2010). Clifford-Holmes et al., (2017a) describes the challenges during participatory processes as follows, "*Within the hybrid approach…modelling is positioned as an activity undertaken in the 'muddled middle', which is conceived as the space between the clean lines of policy design and the messy operational reality of implementation*.". In an attempt to involve a diverse group of stakeholders that represent a variety of interests, follow-up semi-structured interviews were conducted with commercial farmers associated with the study sites (RV3 and GWWA). This was done to capture their framing of perceptions relating to land-use management and to ensure that the relatively low turnout rate would not affect the results of the participatory modelling process. See the summary of the stakeholder engagement schedule in Appendix Table 10.1, which is where the primary data are curated.

***Table 7.1: Participant sector breakdown at multi-stakeholder engagement workshop – Sustainable land-use management in the Middle Berg River Catchment. Sector representation shown: CF = commercial farmer, CP = conservation practitioner, GA = land-use governance authority. Gender representation shown: F = female and M = male.***

| Stakeholder group | Land-use sector representation | Number of participants | | | Gender breakdown of participants |
|---|---|---|---|---|---|
| | | Invited | Confirmed | Attended | |
| Primary | Commercial Farming (CF) | 6 | 6 | 1 | 1 x M |
| | Conservation Practitioner (CP) | 6 | 4 | 3 | 1 x F, 2 x M |
| Secondary | Governance Authority (GA) | 9 | 3 | 3 | 1 x F, 2 x M |
| | **Total** | **21** | **13** | **7** | **2 x F, 5 x M** |

Despite the lower-than-expected turnout, the objectives of the workshop were achieved. However, participants agreed that the underrepresentation of stakeholder, particularly farmers from the commercial agriculture sector, is problematic. **Conservation Practitioner (CP):** "*Try and get more farmers involved, they need to give more input*." And **Government Authority (GA):** "*Research (results) should be shared in a simplified manner, especially for farmers*." Feedback from stakeholders on learning outcomes were captured in a post-workshop survey and further noteworthy comments are indicated in Figure 7.3. The feedback could also be used to inform an iteration of this research by reflecting on which aspects of the stakeholder engagement process and workshop was useful and could therefore be extended upon (e.g. World Café/Sustainability Dialogues and Table Hosts), as well as which parts were difficult (e.g. low turnout rate and Dialogue 3’s discussion on connections) and could be approved upon.


***Figure 7.2: Photo grid of multi-stakeholder participation and materials used in the multi-stakeholder engagement workshop – designed for co-creation and reflection on sustainable land-use management in the Middle Berg River Catchment.***

"*It again emphasized how* ***everything is interrelated***" (CP-M)

"*The positive and negative* ***effects of our human actions*** *on both conservation and farming communities*" (CP-M)

"*How* ***palaeoecology*** *is brought* ***into management***" (CP-F)

"***System thresholds*** *are significant in protecting resources for* ***ES rehabilitation and protection***" (GA-M)

"*There's a* ***lack of collaboration*** *with relevant stakeholders*" (GA-F)

"*It was a* ***good networking*** *session and there are* ***available resources*** *for conservation agriculture*" (CF-M)

***Figure 7.3: Stakeholder perceptions on relevance and significance of the multi-stakeholder engagement workshop.***

Cultivating appropriate accountability with stakeholders is required to provide scientifically rigorous and nuanced decision-support for land-use managers within the Middle Berg River Catchment of the CFR. Therefore, documenting how stakeholders perceive changes of ecosystem services and drivers and the interactions between them in the landscapes they work signifies that their local knowledge is valued as the multiple knowledge systems converge (Jackson et al., 2009; Oteros-Rozas et al., 2015; Phillips et al., 2009; Rice et al., 2009; Wollenberg et al., 2000). Figure 7.4 shows several noteworthy quotes which summarise possible problem definitions that could be explored in further research in the CFR: (a) Plant biodiversity is threatened due to multiple confounding factors. (b) Land-use management decisions heavily influence the level of grazing and there is little long-term information on its impact. (c) Fire regimes are changing and management does not have the adaptive capacity to evolve appropriately. Moreover, it should be noted that significant environmental events such as droughts and wildfires could frame the kinds of discussions during semi-structured interviews and the multi-stakeholder engagement workshop. For example, the 'Cape Town Day Zero' scenario, which government warned as a possibility if dams reached a critical point lower than 18% of their total capacity in May 2018 (LaVanchy et al., 2019; Pascale et al., 2020), took place about a year before the multi-stakeholder workshop on 4 July 2019. Semi-structured interviews took place during the 3-year drought period in the Western Cape (2015-2018). Large wildfires were also common occurrence during the dry season but the stakeholder engagement workshop and all semi-structured interviews took place prior to the Cape Town and UCT fire in April 2021, which had a direct impact on this research project (Nordling, 2021; Rebelo and Esler, 2021; van Wilgen and van Wilgen-Bredenkamp, 2021). Furthermore, the stakeholder engagement workshop took place seven months prior to the start of the COVID-19 pandemic. Additional follow-up semi-structured interviews took place in September and October 2020 with UCT's COVID-19 regulations observed.

(a) *"**Protected areas act as sources of biodiversity** (such as plant genetics). Landowners in lowlands have an increasingly important role in supporting ecosystems at landscape level e.g. if rivers are buffered with a 20/30 m buffer zone, each farm has a section of landscape that is preserved. This leads to a **more resilient landscape**."*

*"**Plant biodiversity** is relevant to us because it **attracts pest control/natural predators** (Mielie Bug - Lady Birds; Snout Beatle – Guinea Fowl; Ducks – Snails). This saves on the use of pesticides (weed and pest control), **leads to better products** and possibility to move towards organic farming."*

(b) *"Need to consider how the loss of large herbivores (such as buck) may have impacted the system? **Loss of these herbivores has likely given rise to long periods of stability, which leads to a loss of biodiversity**. Systems do need some measure of disturbance…we can't just rely on fires to be the only disturbance."*

*"**Game farming** is picking up in the Western Cape, **for economic reasons rather than conservation reasons** - e.g. breeding quagga & different colour morphs of buck/buffalo."*

(c) *"**Fire is more of a driver than herbivory** will ever be since the browse component in fynbos is not favourable or palatable."*

*"**Prevention and awareness is needed around burning** in unfavourable conditions. More environmental awareness is needed in the country. Need awareness for fire seasons in the province. **How much work is required to change behaviour**?"*

***Figure 7.4: Stakeholder narratives of change related to problem definitions on (a) plant biodiversity, (b) herbivory and (c) fire within the CFR.***

Multiple land-use managers with local contextual knowledge have a good understanding of the connections between ecosystem services and drivers (Figure 7.4). Stakeholders recognised the importance of biodiversity and that protected areas acts as a source of biodiversity needed for the resilience and socio-economic benefits. However, the problem is that due to multiple interacting effect such as too frequent fires, damaged topsoil and IAPs, plant biodiversity is likely to decrease sooner than expected as evident in the Elandsberg PNR regime shift in ca. AD 1950s. Stakeholders mentioned that game farming is increasing in the region (such as the reintroduction of large indigenous herbivores at Elandsberg PNR), but this is for economic and not conservation purposes. Stakeholders raised systemic issues around fire and the need for fire public awareness raising, as well as training on how to control and prevent fires. They also mentioned the lack of systems that adequately monitor the effects fire has on biodiversity since the true impacts are unknown.

Furthermore, an extensive list of variables that were elicited as a result of reflection and co-creation during the multi-stakeholder engagement workshop (Process Step 5, Table 4.5) were clustered according to the Iceberg Model (Process Step 6, Table 4.5) as an adaptation of a thematic analysis that was important for problem familiarization (Figure 7.5). This process was a useful part of phase 1 of the SD modelling process to holistically conceptualise the sustainable land-use problem within the SES by illustrating variables and connections using a multi-level systems perspective (Monat and Gannon 2015). The multi-level systems perspective is abbreviated as follows: 'E' = Events/Consequences; 'P' = patterns/behaviours; 'SS' = systemic structures; and 'MM' = mental models/mindsets/physical or chemical forces (see section 2.3).

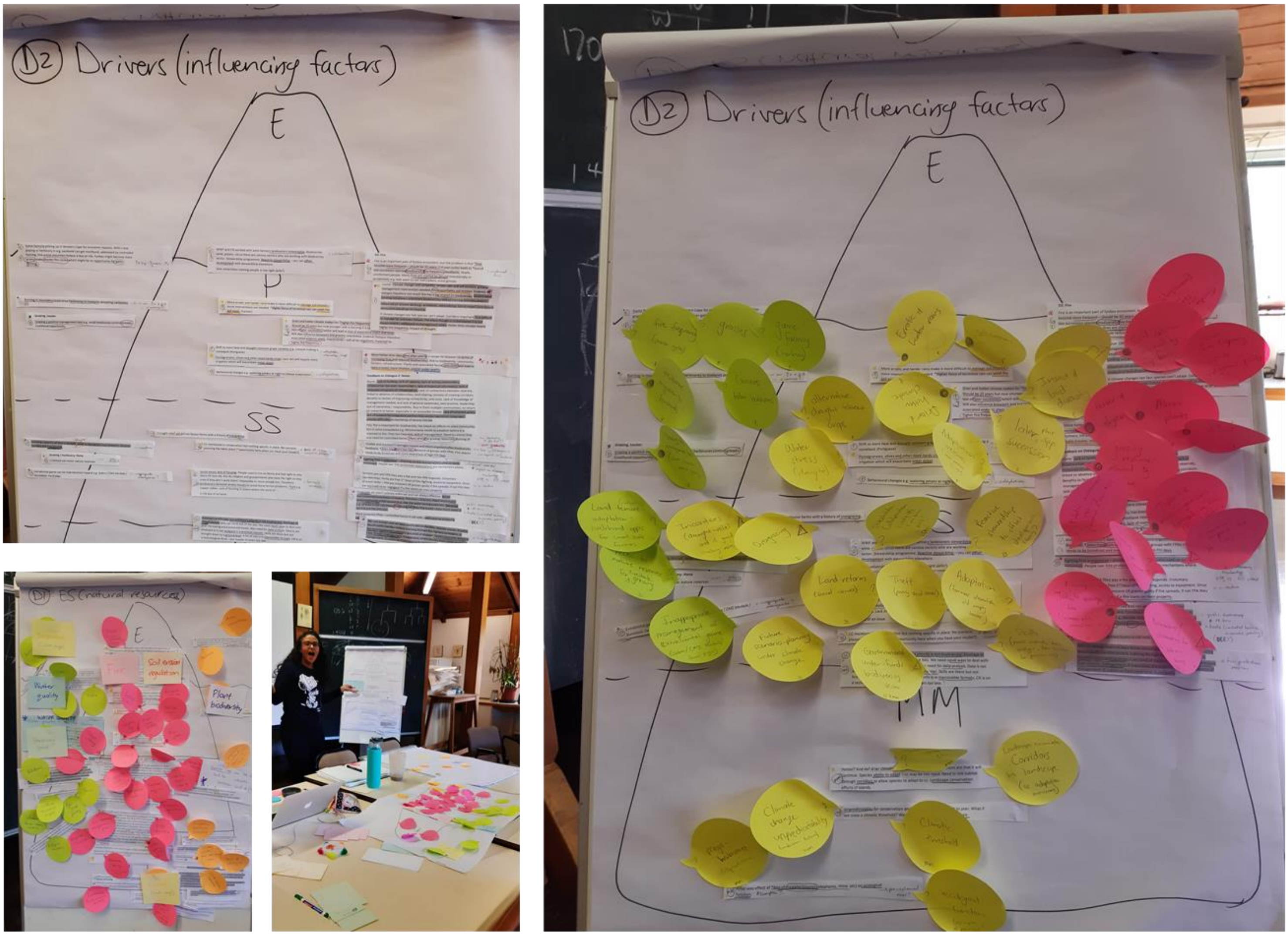



***Figure 7.5: Photo grid of insights from the multi-stakeholder engagement workshop used in a thematic analysis using the Iceberg Model to understand the various systems levels and enhance SES problem familiarization, definition and boundary selection.***

Results from this thematic analysis show that stakeholders most commonly discussed climate change, fire and herbivory (Dialogue 2, Appendix Table 10.2) related to ‘P’ and ‘SS, with some categorized under ‘MM’ as variables related to the unpredictability of climate change adaptation and ecological function. However, stakeholders referred to the recent Cape drought as ‘E’, though this variable related to ‘SS’ that offered drought relief as government incentives to commercial farmers for appropriate stocking rates, as well as collaborations between sectors. Therefore, ‘SS’ can be both physical and intangible, for example, a climatic event such as the drought (E) triggered a change in systemic structure (SS) in the wine agriculture sector. Stakeholder collaboration between farmers and conservation organisations (World Wildlife Foundation (WWF) and CapeNature) could therefore use reactive stewardship to offset negative development impacts from typical intensive agricultural practices.

The multi-stakeholder engagement workshop documented stakeholders' perceptions of change which are regarded as implicit mental models (e.g. Figure 5.12, Figure 6.16 and Figure 7.4). the quotes thematically highlight the nuanced perspectives from multiple stakeholders on the ES and drivers (e.g. mental models on fire and water quality - Figure 5.12 and Figure 6.16, respectively) that are connected to systemic governance variables. Therefore, this finding shows that there was a notable emphasis on variables that were at the SS level that could be leverage points for achieving sustainable land-use management in the region. Some examples include funding of the biodiversity section, illegal clearing of natural vegetation, tools and training to manage fire, rehabilitation of the riparian zone. Connections between the systemic governance issues and the upper levels of the Iceberg Model can be seen in the CLD below (see hexagons as the SS in Figure 7.8) and were captured in detail in the systems map (Appendix Figure 10.24). By using the stakeholder perceptions of the connections between ecosystem services and drivers (Dialogue 3, Appendix Table 10.2), the SES variables were illustrated via a systems map as per Process Step 7 in Table 4.5. The systems map (Appendix Figure 10.24) aided in zooming in on a particular SES problem and boundary selection (i.e. identifying key variables and stocks that would be modelled), which is discussed in section 7.2.2 below. Multi-stakeholders' perceptions were made more explicit by recording their mental models in the systems map and during subsequent qualitive (CLD) and quantitative (SFD) model development.

#### *7.1.2.2 The SES problem at Elandsberg PNR*

Little is known about Renosterbos' historic abundance and distribution, leaving managers and conservationists to speculate on whether current vegetation composition of Renosterveld is typical, should be regarded as "degraded Grasslands" or "degraded Fynbos" and therefore whether Renosterveld landscapes are a worthy conservation target and restoration efforts are needed (sections 2.2.2.2 and 4.1.3.2). The central modelling problem of the Elandsberg PNR the case study is how to increase plant biodiversity and therefore improve ecosystem function for plant biodiversity conservation and possibly sustainable ecotourism practices in the future. Together with the long-term patterns from high temporal resolution, multi-proxy palaeo-data as reference modes (Process Step 8, Table 4.5) connections with systemic structure and mental models, as derived from stakeholder engagement insights, become more explicit.

Mapping out the SES problem in this way is useful in identification of potential leverage points that could be activated for change. These leverage points are places to intervene in the system that influence the feedback loops of ecosystem services and drivers that cause the SES problems described by stakeholders. In the Elandsberg PNR case, palaeoecological data from a ca. 1300-year-old VANG sedimentary core (Forbes et al., 2018) and large stock unit (LSU) data were used as reference modes (Figure 7.6 and Figure 7.7) for the SD simulation model. The VANG core was extracted from a small sized wetland on an ecotone between Swartland Shale Renosterveld and Swartland Alluvium Fynbos (Mucina and Rutherford, 2006) (Figure 3.4), which makes it useful in detecting vegetation change because species are at their biological and/or environmental limits at the between-biome scale (Gillson et al., 2020; MacPherson et al., 2019). Thus, shifts between alternative stable states and critical ecological thresholds are easier to identify and therefore a point where one vegetation state reaches a threshold and reorganises into a different vegetation assemblage that is more detectable (Gillson and Ekblom, 2009; Gunderson and Holling, 2002). Therefore, long-term data can provide clues into what patterns are associated with system structure and modelling the complex SES problem related to land degradation and associated biodiversity loss will help to gain insights to improve biodiversity conservation at this site.

Fossil pollen from the most abundant pollen types in the VANG palaeo-record (Figure 7.7) is used as a proxy for plant biodiversity: Asteraceae long/high-spine type-1 pollen as a pollen type which makes up multiple Asteraceous plant species but is indistinguishable at the family and genus level; and Asteraceae *Stoebe/Elytropappus*-type pollen (hereafter, Renosterbos pollen) which is a single Asteraceous species, Renosterbos, that is unpalatable and when in high abundances is considered less desirable for conservation. Palaeo-proxies for drivers of change associated with land-use disturbance include: Macro-charcoal as a proxy for local fire history and coprophilous fungal spores as a proxy for herbivory/grazing. Palaeoecological results show two ecological regimes (domains of attraction or alternative stable states) occurring over the past ca.1300 years. The regime shift occurred in the ca. AD 1950s and it is characterised by a decrease in plant biodiversity, evident by a decrease in Asteraceous pollen and an unprecedented increase in Renosterbos pollen in the time period in the sediment sequence from this site (Figure 7.7). Decreased plant biodiversity was driven by increased grazing since ca. AD 1800s and later the interacting effects between increased grazing and burning caused by agricultural intensification. Trends continued during the AD 1970s (when Elandsberg PNR was proclaimed as a nature reserve) until the present.

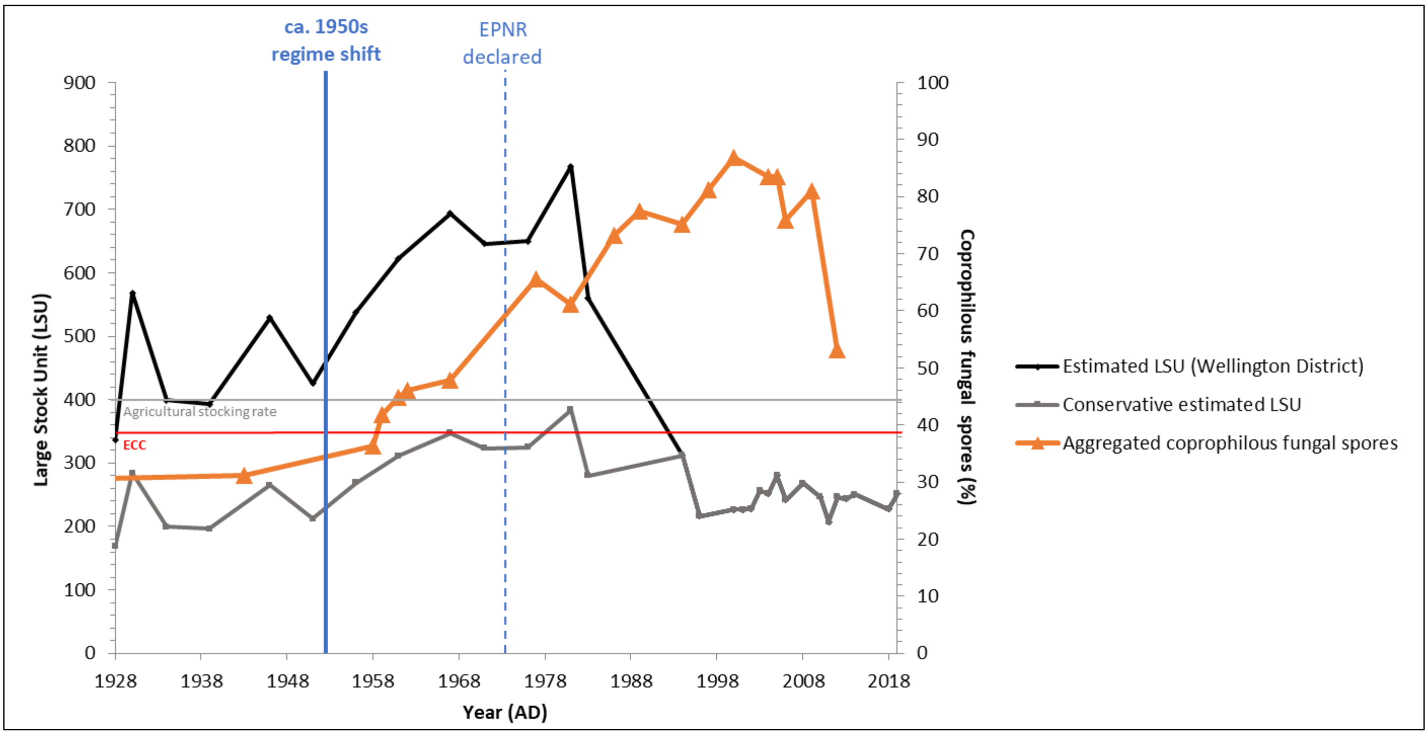



***Figure 7.6. Trends in Large Stock Unit (LSU) data compared with herbivory data (fossil coprophilous fungal spores) from the VANG core, Elandsberg PNR*** (Forbes et al., 2018)***. Given the limited LSU historical records (AD 1994-2019, Game Management Plan, Wooding pers. comm. 2020), LSU data was extrapolated for the period AD 1928-1994 using estimated stocking rates (LSU/ha) from Wellington magisterial district records as reported in the Agricultural Censuses (e.g. Statistics South Africa, 2010) for commercial farming areas. Increasing trends are observed in both the LSU and the spore data until AD 1981. Red horizontal line shows the recommended ecological carrying capacity (ECC) and the grey horizontal line shows the recommended agricultural stocking rate. The solid blue vertical line shows the timing of the ecological regime shift, while the dashed line shows when the site was declared a nature reserved in AD 1973***

Traditional formats of palaeo-proxy diagrams included in scientific articles of peer-reviewed journals such as Figure 7.7 could be perceived as too technical for anyone outside the field and particularly key local stakeholders, and may therefore be ineffective and not beneficial for on the ground impact. Therefore, Storylines and Scenarios of the Elandsberg PNR case study (Figure 7.8) provides an example of how the conceptual meta-framework can be used and is an end-product that may be user-friendly for stakeholders. The rationale for compiling the Storylines and Scenarios is explained in section 4.2.1. The long-term palaeo-perspective provides patterns of the past and thus the quantitative evidence-base while stakeholder narratives of change enhance the story of the present by providing a process-based understanding of why SESs have changed and the role that current, governance and practice plays in influencing this. The CLD in Figure 7.8 is a representation of the systems map (Figure 10.24). It demonstrates how expert input from commercial farmers, conservation practitioners and government authorities define the "problem space" that details the causal explanations on a generic and less technical level within the realm of environmental decision-making processes (Videira et al., 2010). For example, "erratic and harder rain" could be quantified as rainfall frequency and intensity and "tools and training to manage fire" is related to controlled burns and quantifying fire return intervals, fire intensity and fuel accumulation.

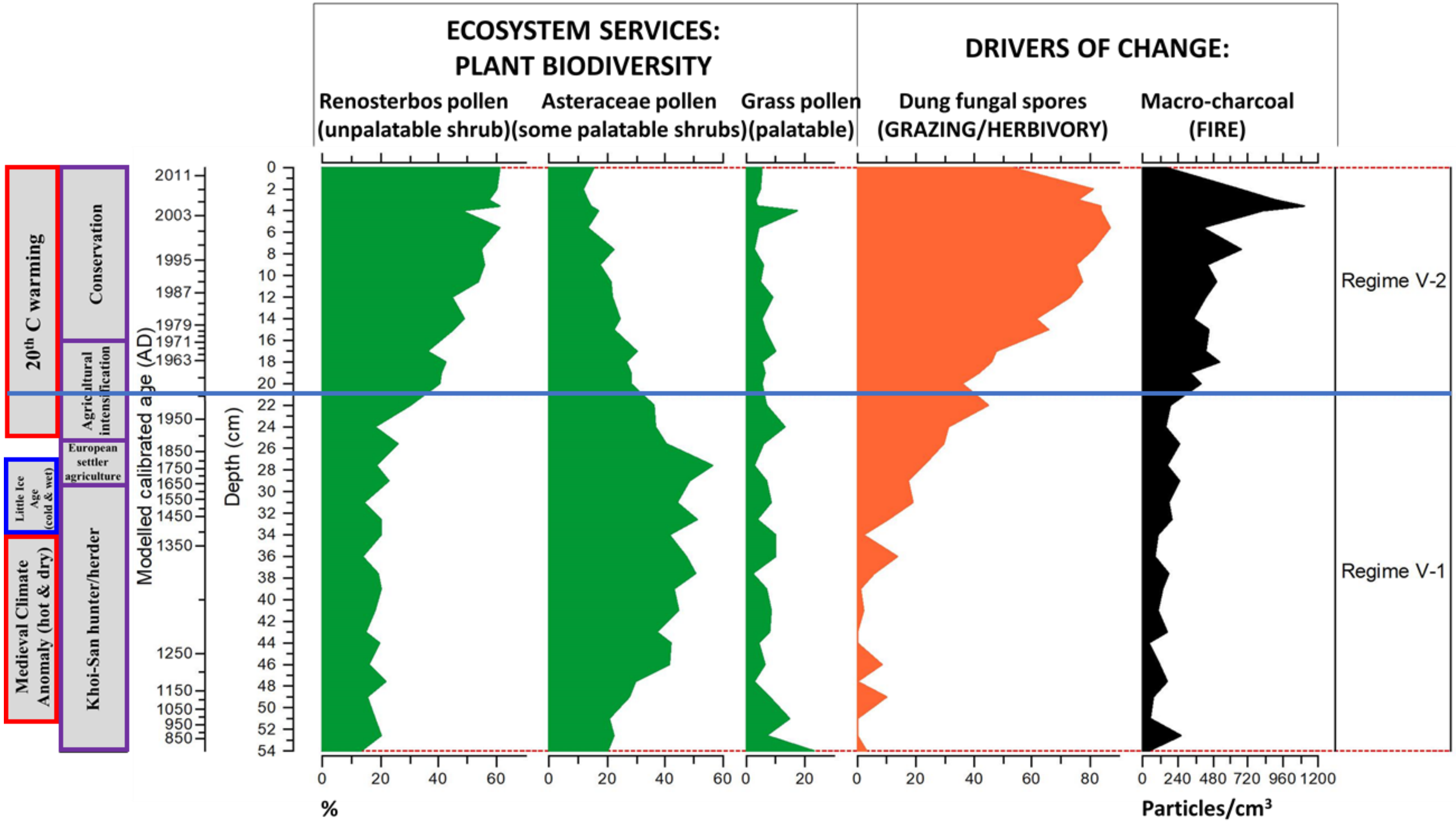



*Figure 7.7: Synthesis diagram of selected palaeoecological proxy data from Elandsberg Private Nature Reserve, South Africa used as reference modes that show problematic behaviour over time: decreased plant biodiversity (pollen), and increased grazing (coprophilous/dung fungal spores) and local fire (macro-charcoal). Climate variation and land-use during the ca.1300 years is summarised on the far left. Pollen zones depict a regime shift: Zone V-1 is <1950s and Zone V-2 is 1950s-2012.*

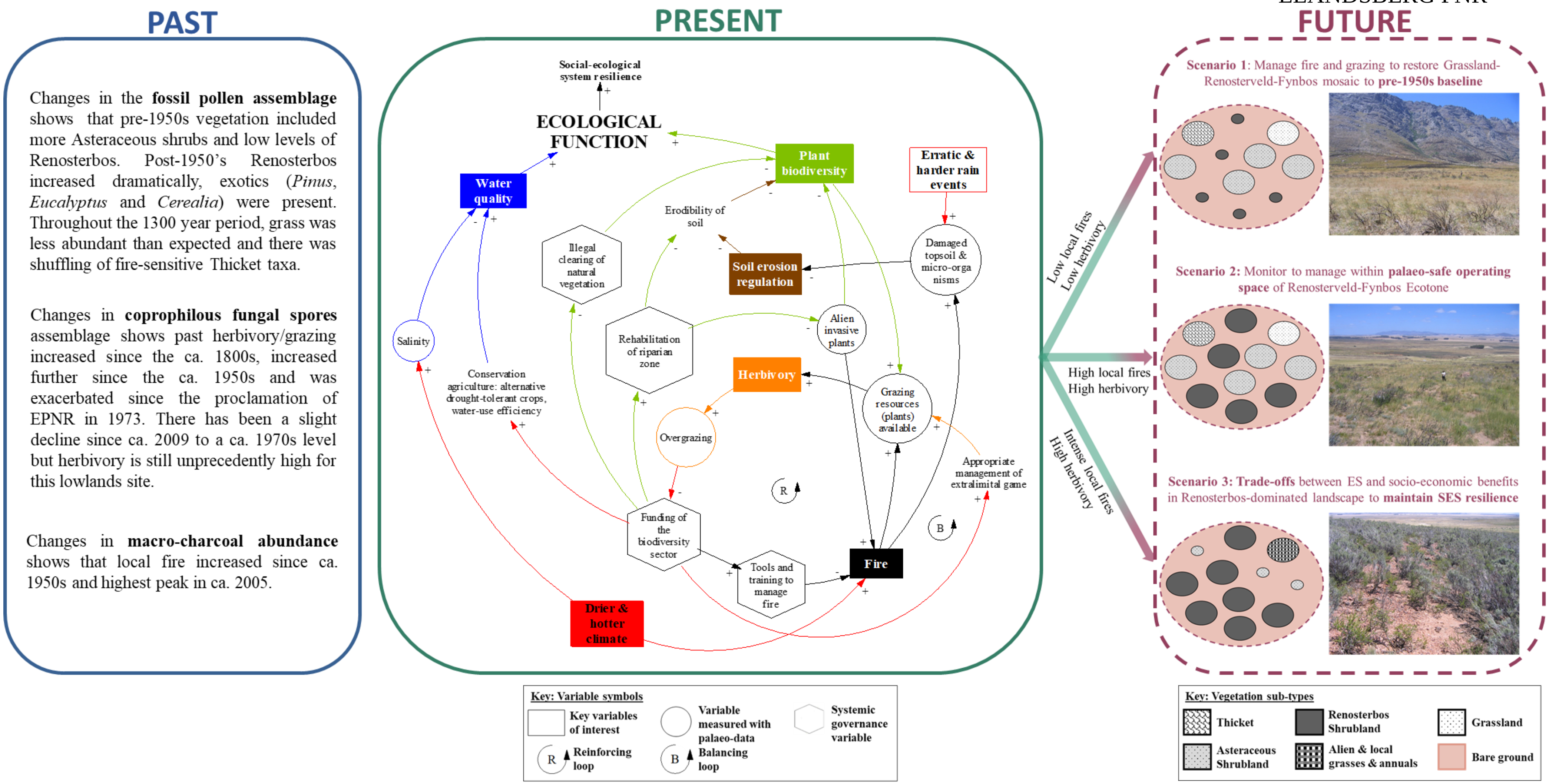



*Figure 7.8: Synthesis of Storylines and Scenarios for Elandsberg PNR that applies the past-present-future lens for environmental change (see section 4.2.1). The past captures the long-term patterns. The CLD summarises stakeholder narratives of change in the present. Three potential policy/management scenarios are described for the future.*

Moreover, to make the evidence- and process-based understanding even more prescriptive in nature, potential future scenarios for management (section 2.4.2.2) are presented in the far right of Figure 7.8. This is a way for stakeholders to visualise what the future landscape may look like under various management strategies including, restoration (best-case Scenario 1); monitoring (Scenario 2); or trade-offs between multiple ecosystem services (Scenario 3). An example of trade-offs could be. managing for endemism and plant biodiversity conservation versus managing for ecotourism, game meat production and other recreational services (Holden et al., 2021). Scenario 3 could hypothetically become the worst-case scenario if a critical ecological threshold is crossed in the future. The critical transition to an alternative stable state, namely 'Degraded Renosterveld' dominated by large homogenous patches of Renosterbos could occur if burning and grazing continues to increase (Forbes et al., 2018). Increased fire risk could be exacerbated by climate change (Van Wilgen et al., 2010).

## 7.2 PSD Theoretical Framework – model formulation and testing

### 7.2.1 Dynamic hypothesis

By focusing on a selected aspect of the CLD depicted in Figure 7.1 above, ecological integrity was expanded in the case-specific CLD shown in Figure 7.9. This case-specific CLD was important for model boundary selection (Process Step 9, Table 4.5). Ecological integrity is explored through the changes in plant biodiversity (provisioning/supporting ecosystem service). It illustrates the important causal relationships, including six feedback loops (R1-R4 and B1 and B2), which underpin the systemic structure of the SES problem. Factors mentioned in the *Asteraceous shrubs dominance* (R1) and *Renosterbos dominance* (R2) loops are the most central to the dynamic behaviour over time. A shift in loop dominance from *Asteraceous shrubs dominance* (R1) during the pre-1950s to *Renosterbos dominance* (R2) post-1950s is explained by the threshold behaviour observed in the palaeo-record between ca.AD 1800-1950 (Figure 7.7) and how the drivers and feedbacks interacted non-linearly and caused the system to shift from one stable state (Regime V-1: Asteraceous shrubs dominated regime) to an alternative stable state (Regime V-2: unpalatable Renosterbos dominated regime) (Figure 7.7)

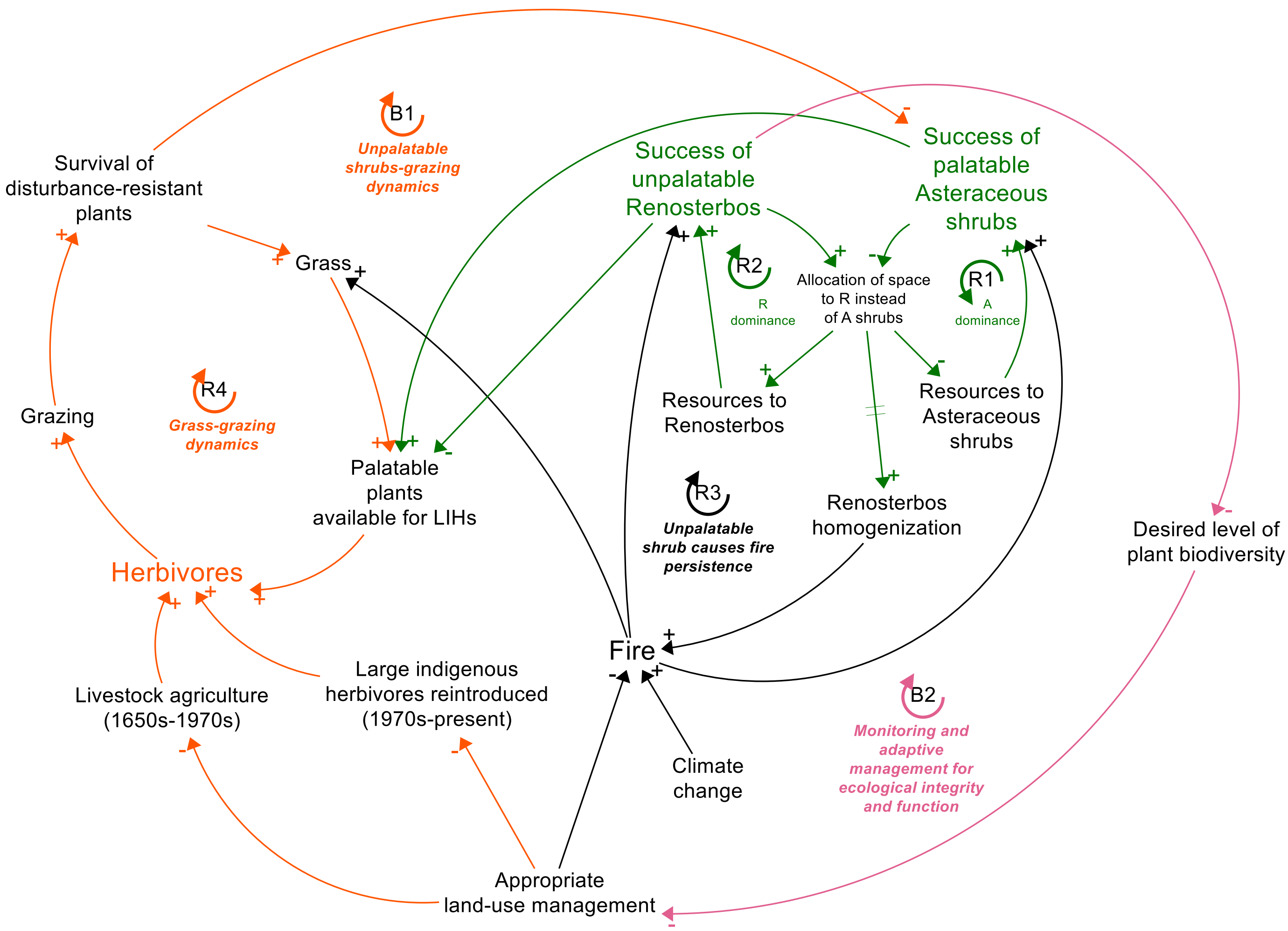



***Figure 7.9: Causal loop diagram depicting the dynamic hypothesis (feedback loops R1-R4 and B1 and B2) for the Ecological Model for biodiversity conservation based on Forbes et al., 2018 and Forbes, 2014. Connectors highlighted in pink influence the management policy and scenario analyses.***

Given the low levels of grazing on palatable plants (grasses and some Asteraceous shrubs) during pre-1950s (Regime V-1), Asteraceous shrubs remained dominant in the Elandsberg PNR landscape. This dominance is reinforced by the feedback between resources and the space available for Asteraceous shrubs (reinforcing loop R1) and Renosterbos (reinforcing loop R2) and low grazing and fire levels (R4 and B1). Regime V-1 in the pollen record is characterised by a heterogeneous, patch-mosaic landscape that includes patches of different vegetation units/elements (i.e. patches of Alluvium Fynbos, Renosterveld-Fynbos Ecotone, Grassland, Shale Renosterveld, Grassland-Renosterveld matrix, and Thicket) that result in less wide-spread fires thus promoting further heterogeneity in vegetation units and varying post-fire ages of vegetation patches (Forbes et al., 2018). However, once grazing levels reached a certain level and crossed an ecological threshold, loop dominance favoured Renosterbos competition. The feedbacks that maintained Regime V-2 are high grazing-fire dynamics (R3, R4 and B1), which reinforces competition that favours the unpalatable Renosterbos (R2) over other palatable Asteraceous shrubs. Despite Renosterbos dominance, at the present day the site remains heterogeneous to a certain extent since it includes patches with different vegetation types/units - Asteraceous shrubs, thicket and indigenous grasses and varying post-fire ages persist in the landscape though at lower levels (Figure 3.5).

The initial hypothesis is as follows: If current levels of grazing by reintroduced large indigenous herbivores are maintained then Renosterbos will outcompete more diverse Asteraceous plant communities. This scenario would be outside of the historical range of variability as indicated by the palaeo-record that shows the historical range of variability for ecosystem health, function and climate resilience at this site over the last ca. 1300 years. With increasing fire, further pressure from anthropogenic-induced climate change will exacerbate Renosterbos spread and overall landscape degradation and homogenisation. The CLD does not detail how appropriate management can influence both fire prevention (decreased fire frequency) and fire control (decreased fire intensity and planned controlled burns), however, this is explored in the SFD. Further justification for the variables and how the feedback loops were hypothesised is supported by model documentation and references shown in Appendix 10.7.2.

The dynamic hypothesis shows that if Regime V-2 trends continue, ecotourism including both botanical conservation and game farming may no longer be a sustainable conservation practice at this site in the future, because the palatable vegetation will be depleted. The main impact on ecosystem services would be a decrease in plant biodiversity (supporting and provisioning services) with subsequent negative impacts on other ecosystem services such as water quality, soil erosion regulation, pollination (regulating services) and recreation, livelihoods, economic activity due to social conflict between biodiversity conservation and ecotourism priorities (cultural services).

### 7.2.2 Model boundary, structure and time horizon

One of the main objectives of the multi-stakeholder engagement workshop was to identify model boundaries early in the SD modelling process (Process Step 5-7, Table 4.5). However, when the workshops participants articulated topics that were not directly related to the selected ecosystem services (plant biodiversity, water quality and soil erosion regulation) and drivers (fire, herbivory and climate change) (dialogue questions in Appendix Table 10.2), participants were encouraged to use the 'parking lot' (
Figure 7.2). This is because such topics are still pertinent to sustainable land-use management within the Middle Berg River Catchment context of the CFR. Towards the end of the workshop, we used a full group discussion to solicit additional variables and reflect on their relevance. The systems map (Appendix Figure 10.24) represents the group consensus on the connections between ecosystem services and drivers and was therefore used as guidance on what SES problem could be modelled and which case study within the focal area along a gradient of elevation and land-use intensity would be feasible to model. The goal tree (Figure 7.10) considers stakeholders' goals and also makes the problem more explicit. Although all variables in the systems map and the goal tree are relevant, they have not all been considered in the SFD presented here and therefore require further development.

A summary of the model structure is shown in Figure 7.11 and the full model structure shown in Figure 7.12. Key variables with initial values and the model decision rules are summarised in Table 7.2 and Table 7.3, respectively. Full model documentation (including historical data and equation breakdown) allows for the model to be fully reproduced (Appendix Table 10.9). In the current model structure, a boundary decision was made to focus on the effects of land-use disturbance (fire and grazing) on Renosterbos and Asteraceous shrub cover, excluding grasses. Although climate change has an indirect effect on grazing (i.e. less precipitation has a direct effect on primary productivity and thus the vegetation cover as a grazing resource) and, such model structure was excluded as another model boundary selection decision. However, climate was considered in the model analysis as per the literature that reports climate warming impacting on the Fire Danger Index (FDI) days and consequently the likelihood of increased fire occurrence (Forsyth and van Wilgen, 2008; Kraaij and Wilgen, 2014) (see Table 7.5 and Appendix Table 10.17).

The time horizon chosen for the model simulation is ca. AD 750-2100, however the scenarios and policies are only simulated from 2020 onwards (i.e. 80 years). Although the regime shift only occurred a few decades ago (ca. AD 1950s) and the time taken leading to the threshold behaviour is centuries. As the temporal scale of the palaeoecological data is large (ca. 1300 years) and modelling for the same amount of time into the future is common practice in SD, considering a longer time horizon until ca. AD 3250 would be considered typical. However, simulating so far into the future creates too much uncertainty. However, modelled results for stakeholders are only discussed from 1950-2100 since this timescale could detect future regime shifts. The time horizon includes the 2030 Sustainable Development Goals (United Nations Development Program (UNDP), 2016) (section 2.5.1) and 2050 timescale used in most climate change projections (Haensler et al., 2011; Meadows, 2006; Midgley and Thuiller, 2007; Midgley et al., 2005; Turpie et al., 2002).

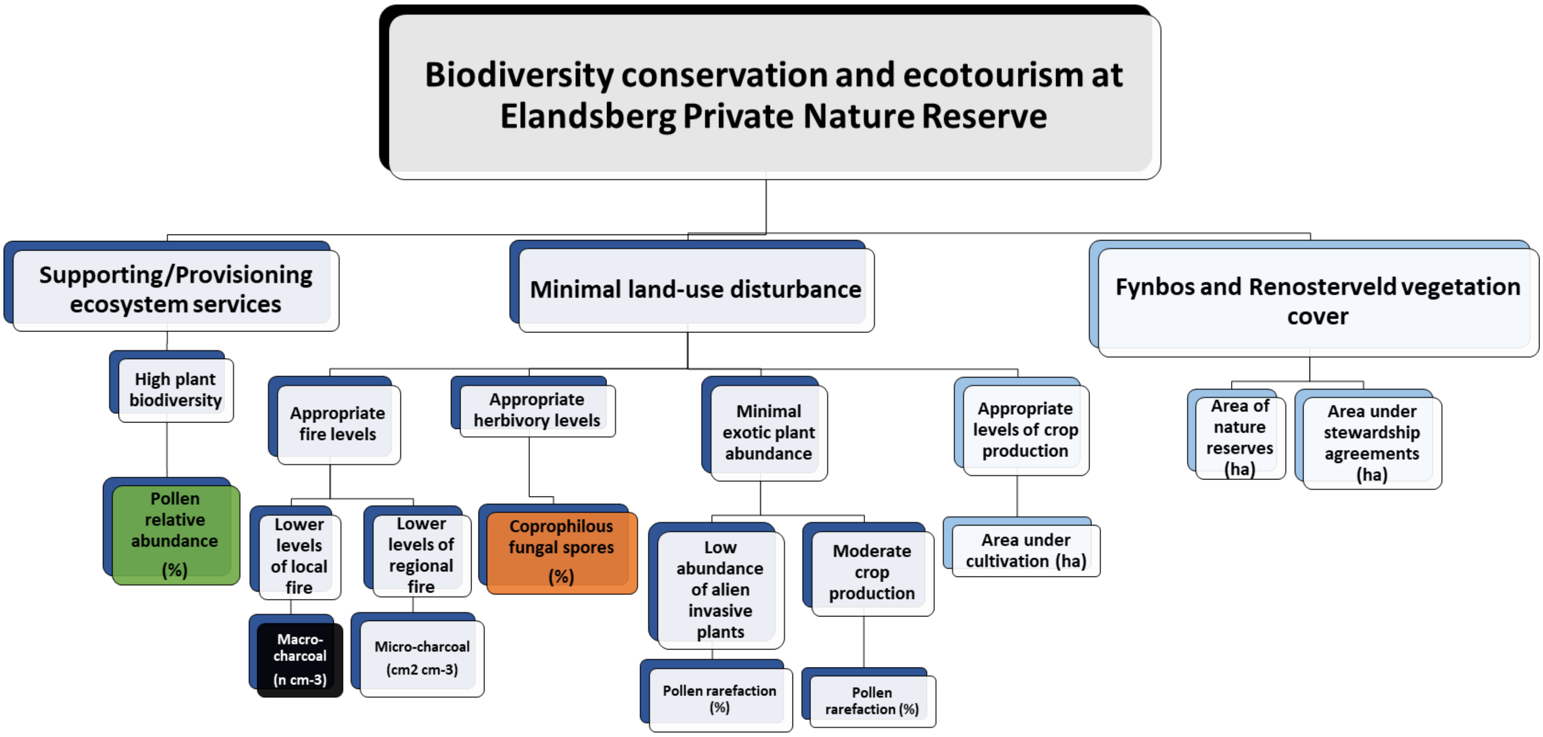



*Figure 7.10: Goal tree defining the model boundary. Variables highlighted in dark blue can be measured directly using palaeoecological data; variables highlighted in light blue cannot be inferred but are important in explaining palaeoecological data; and variables highlighted in green (plant biodiversity) are black (local fire) are endogenous and the variable highlighted in orange (herbivory/grazing) is exogenous in the Ecological Model.*

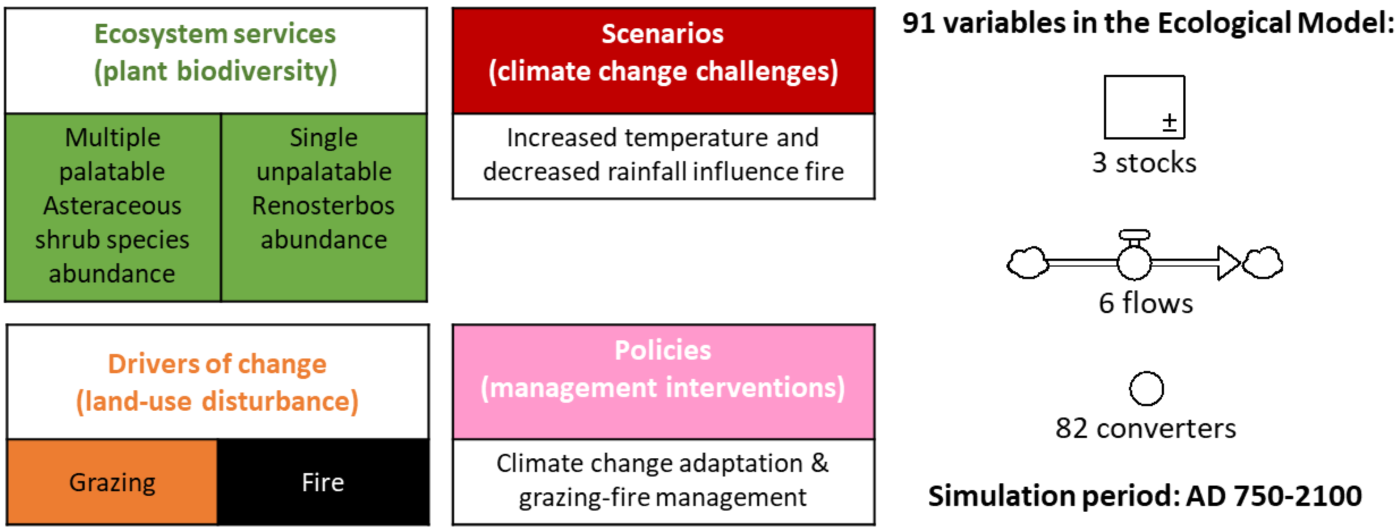



*Figure 7.11: Summary of model structure captured in the stock and flow diagram (SFD) of the Ecological Model.*

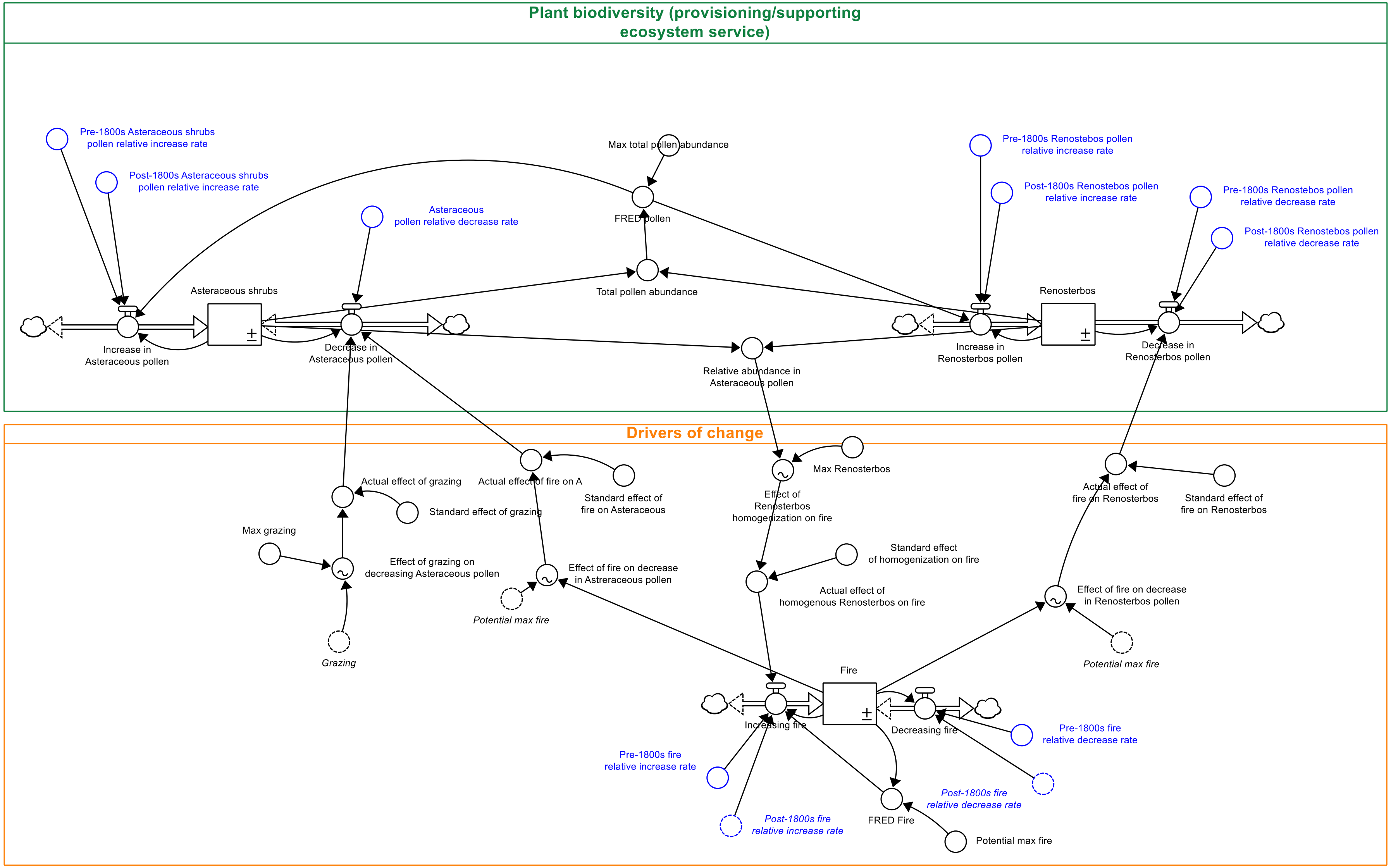



***Figure 7.12: Preliminary stock and flow diagram (SFD) of the Ecological Model for Elandsberg PNR – page 1/2.***

*Ecological Model for Elandsberg PNR - page 2/2*

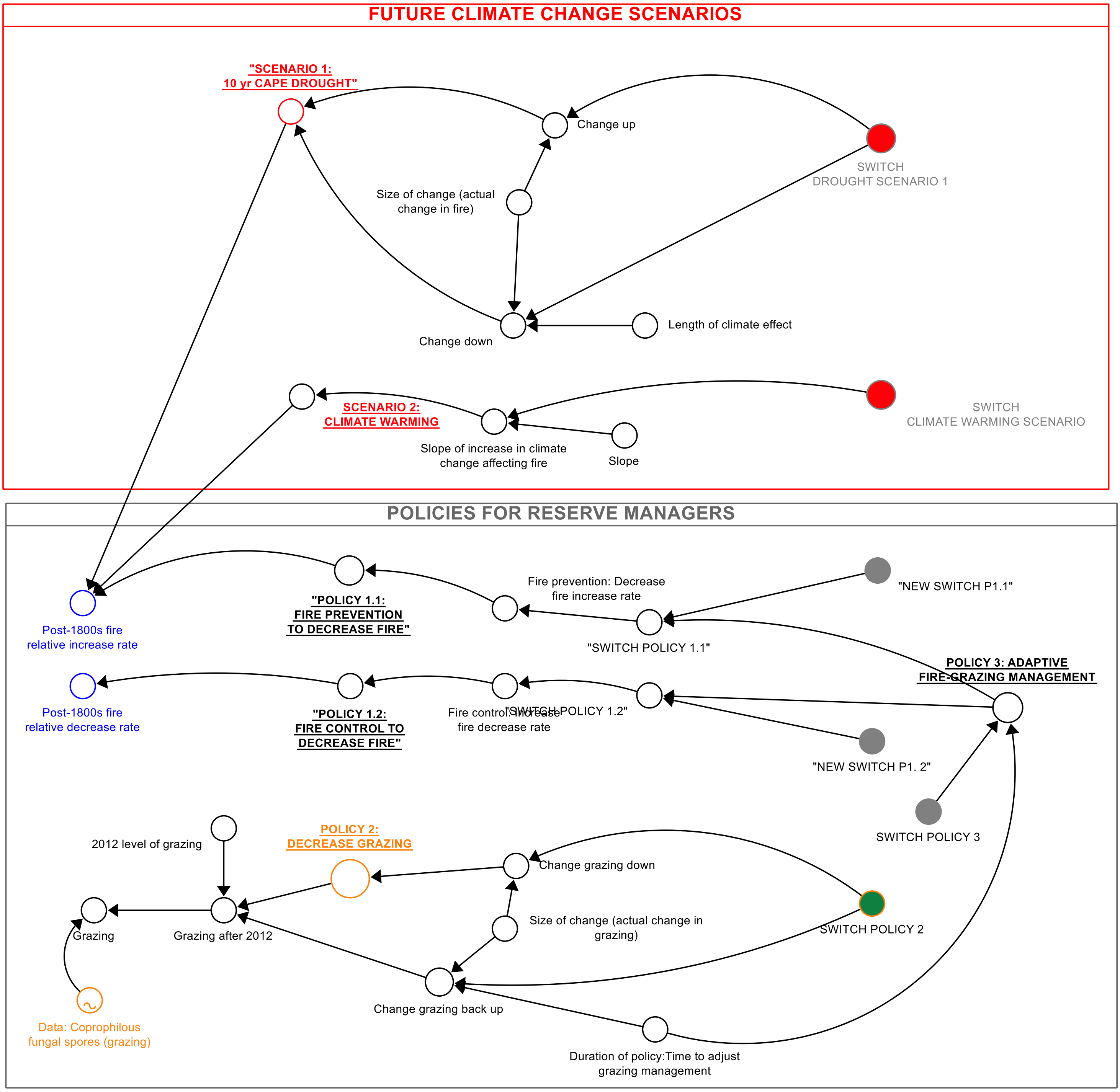
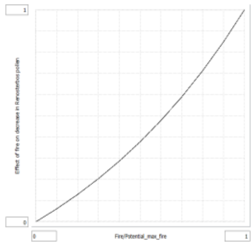
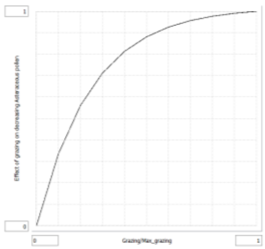
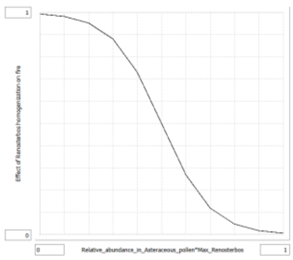


*Table 7.2: Parameters with initial values used in the Ecological Model. Data source: Forbes et al. 2018.*

| Key variable | Variable type | Initial value | Units | Reference, assumption or calculation |
|---|---|---|---|---|
| **Max total pollen abundance** | Constant (exogenous) | 0.77 | 1 | **Calculated:** This is the maximum total abundance of Asteraceous shrubs pollen plus Renosterbos pollen (77%) in the palaeoecological record. The remainder (33%) is made up of other pollen taxa not included in this model (Forbes et al. 2018). |
| **Asteraceous shrub pollen**<br>*Equation: Asteraceous shrubs pollen(t) = Asteraceous shrubs pollen(t - dt) + (Increase in Asteraceous pollen - Decrease in Asteraceous pollen) * dt* | Stock (endogenous) | 0.20 | 1 | **Existing literature:** Initial value is 0.2 as per data from Forbes et al. (2018). Unit is dimensionless since it is pollen proportion (percentage) data. |
| **Asteraceous pollen relative increase rate** | Constant (exogenous) | 0.0159 | 1/Year | **Assumption** that this rate would be similar to the decrease rate. |
| **Asteraceous pollen relative decrease rate** | Constant (exogenous ) | 0.0159 | 1/Year | **Calculated:** Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. There was no need to have different rates for pre- and post-1800s in this case since the optimization revealed similar rates. |
| **Max Renosterbos** | Constant (exogenous ) | 0.61 | 1 | **Existing literature:** This is the maximum observed Renosterbos pollen in the palaeo record from this site (Forbes et al.2018). It is % or proportion data. |
| **Renosterbos pollen**<br>*Equation: Renosterbos_pollen(t) = Renosterbos_pollen(t - dt) + (Increase_in_Renosterbos_pollen - Decrease_in_Renosterbos_pollen) * dt* | Stock (endogenous) | 0.13 | 1 | **Existing literature:** Initial value is 0.13 as per data from Forbes et al. (2018). Unit is dimensionless since it is pollen proportion (percentage) data. |
| **Pre-1800s Renostebos pollen relative increase rate** | Constant (exogenous) | 0.0225 | 1/Year | **Calculated:** Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| **Post-1800s Renostebos pollen relative increase rate** | Constant (exogenous ) | 0.2046 | 1/Year | |
| **Pre-1800s Renostebos pollen relative decrease rate** | Constant (exogenous) | 0.2340 | 1/Year | |
| **Post-1800s Renostebos pollen relative decrease rate** | Constant (exogenous) | 0.2407 | 1/Year | |
| **Standard effect of homogenization on fire** | Constant (exogenous) | 1.1432 | 1 | **Calculated** using optimization in Stella: Based on reliable palaeo data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate effect. |
| **Potential max fire** | Constant (exogenous) | 2200 | n cm^-3 | **Assumption** that potential max fire can be double (2200 n cm^-3) the observed max fire (1100 n cm^-3). Optimization in Stella. |
| **Fire**<br>*Equation: Fire(t) = Fire(t - dt) + (Increasing fire - Decreasing fire) * dt* | Stock (endogenous) | 74 | n cm^-3 | **Existing literature:** Initial value as per data from Forbes et al. (2018). Unit is number of macro-charcoal particles per cubic cm. |
| **Pre-1800s fire relative increase rate** | Constant (exogenous) | 0.3636 | 1/Year | **Calculated:** Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| **Post-1800s fire relative increase rate** | Constant (exogenous) | 0.2292 | 1/Year | |
| **Pre-1800s fire relative decrease rate** | Constant (exogenous) | 0.1009 | 1/Year | |
| **Post-1800s fire relative decrease rate** | Constant (exogenous) | 0.0517 | 1/Year | |
| **Standard effect of fire on Asteraceous** | Constant (exogenous) | 1.0249 | 1 | **Calculated** using optimization in Stella: Based on reliable palaeo data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate effect. |
| **Standard effect of fire on Renosterbos** | Constant (exogenous) | 0.9998 | 1 | |
| **Data: Grazing** | Constant (exogenous) | GRAPH(TIME) | 1 | **Existing literature:** Grazing is currently an exogenous variable. Dung fungal spore data from 751 to 2012 (Forbes et al. 2018). |
| **2012 level of grazing** | Constant (exogenous) | 0.53 | 1 | **Existing literature:** 2012 level of dung fungal spores is 53% (Forbes et al. 2018). |
| **Max grazing** | Constant (exogenous) | 0.86 | 1 | **Existing literature:** This is the maximum observed dung fungal spores in the palaeo record from this site (Forbes et al.2018). It is % or proportion data. |
| **Standard effect of grazing** | Constant (exogenous) | 2.6455 | 1 | **Calculated** using optimization in Stella: Based on reliable palaeo data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate effect. |
| **Standard effect of grazing on Asteraceae increase** | Constant (exogenous) | 2.6455 | 1 | |

***Table 7.3: Decision rules related to ecological dynamics, policy (management interventions) and scenarios (climate as an environmental challenge and influencing factor) used in the Ecological Model.***

| | Decision rule | Decision point | Description |
|---|---|---|---|
| **Ecological dynamics** | Effect of fire on decrease in Astreraceous pollen; and Effect of fire on decrease in Renosterbos pollen | **Standard effect of fire on Asteraceous;** and **Standard effect of fire on Renosterbos;** | Assumption that the effect will be that of exponential growth. Renosterveld taxa (Renosterbos and other Asteraceous shrubs) are fire-prone and fire-adapted (Bond 1980; van Wilgen. 1982; van Wilgen et al. 2012).  |
| | Effect of grazing on decreasing Asteraceous pollen | **Standard effect of grazing** | Asteraceous shrubs decrease since they are palatable for herbivores. Grazing directly affects the decrease rate of Asteraceous pollen since herbivores will mostly eat younger more palatable plants. More grazing will occur but herbivore population will eventually reach a carrying capacity since pastures become less available over time.  |
| | Effect of Renosterbos homogenization on fire | **Relative abundance in Asteraceous pollen** | This is comparing the relative abundance of Asteraceous shrubs and Renosterbos to determine whether there is a well-balanced ratio or if Renosterbos is dominant and therefore having a homogenization effect. *Equation: Relative abundance in Asteraceous pollen = (MIN(Asteraceous shrubs pollen/Renosterbos pollen; 1))* |
| | | **Standard effect of homogenization on fire** | Since a patch-mosaic landscape if preferable and the variety in vegetation units can act as natural fire refugia (e.g. thicket by boulders) (Forbes et al. 2018). The timing of fires (i.e. burn season in autumn versus spring; Levyns 1935; Cowling et al. 1986) and grazing (i.e. immediately after a burn or not; Rebelo 1995) are important factors that will effect biodiversity and patch structure and therefore burn differently when a fire comes through. Assumption is the effect is a S-shaped decrease.  |

*(Decision rules table continued)*

| | Decision rule | Decision point | Description |
|---|---|---|---|
| **Policy 1.1: Fire prevention** | STEP(Fire prevention: Decrease fire increase rate; 2020) | **Fire prevention: Decrease fire increase rate** | Stakeholders' suggestion to prevent fire from taking place by having awareness-raising and fire alerts. (Reference: Social Learning Workshop on 4 July 2019). Assumption: 0.1 rate to decrease fire per year. |
| **Policy 1.2: Fire control** | STEP(Fire control: Increase fire decrease rate; 2020) | **Fire control: Increase fire decrease rate** | Stakeholders' suggestion to improve fire control have enforcing fire break laws and improving Fire Support Association initiatives such as training during controlled burns (Reference: Workshop 1). Assumption: 0.1 rate to decrease fire per year. |
| **Policy 2: Decrease to 9.5% grazing only** | STEP(Change grazing down; 2020) | **Duration of policy:Time to adjust grazing management (5-50 years)** | Policy 2 is the decision taken by Elandsberg PNR to reduce their stocking number to the pre-1950s level (9.35%) (Forbes et al. 2018) for certain period of time (5, 10 or 50 years) while the grazing reduction policy is in place before it can return to its previous level. |
| | | **Change grazing down;** and **Change grazing back up** | The safe operating space is between zero grazing to the max level of grazing before the regime shift in 1950s, therefore 0-31%. Ideally grazing would be at the pre-1950s average of 9.35%. Therefore, 2012 level (53%) minus desired level (9.35%): 53-9.35 = 43.65% (0.4356). Policy sensitivity needs to be done to determine how long this policy needs to be in place before managers can increase the stock again. |
| **Policy 3: Adaptive grazing-fire management** | (1-SWITCH_POLICY_3)*STEP(0; 2012+Duration of policy:Time to adjust grazing management)+(SWITCH POLICY 3)*STEP(1; 2012+Duration of policy:Time to adjust grazing management) | (combination of above decision points) | Policy 3 is Adaptive grazing-fire management where Elandsberg PNR managers decrease grazing to the average pre-1950s levels (9.35%) for a certain period of time (5, 10 or 50 years) and then also implement management that aims to decrease fire (via fire prevention - Policy 1.1 and control - Policy 1.2). |

### 7.2.3 General model calibration and assumptions

The principles for modelling decision-making are two-fold: The structure of the model is based on assumptions about the (i) physical and institutional environment (i.e. capturing the structure of the SES); and about the (ii) decision process of key stakeholders who operate in the real system (i.e. influenced by the mental models underlying the structure). The Ecological Model was calibrated using selected palaeo-proxies from Elandsberg PNR dated ca. AD 750-2012 (Forbes et al., 2018). Additionally, qualitative multi-stakeholder engagement information from the multi-stakeholder engagement workshop (held on 4 July 2019), with participants represented multiple land-use management sectors (conservation practitioners (3), commercial farmer (1), government authorities (3)) were consulted. Stakeholder engagement provided context for perceptions of past and future drivers of change (i.e. land-use and climate change) thus providing a values-based interpretation of the changes in ecosystem services and drivers, given the principles-based concept of resilience and justice (Hermans and Knippenberg, 2006; Pope et al., 2017). The term mental model is viewed as “a relatively enduring and accessible, but limited, internal conceptual representation of an external system (historical, existing or projected) whose structure is analogous to the perceived structure of that system" (Doyle and Ford, 1999, 1998). In this context, stakeholders could share their mental models (i.e. tacit knowledge) via social learning techniques, which was then captured in the systems map and CLDs. Together, palaeoecological data and insights from the systems map was used for the development of a dynamic hypothesis and model analyses. Therefore, mental models are now being converted to explicit knowledge that other stakeholders can access (Kopainsky et al., 2017; Sims and Sinclair, 2008). Assumptions based on the reliability of the palaeo-data and the vegetation dynamics literature allowed for optimisation of rates that influenced the main flows (increasing and decreasing Asteraceous, Renosterbos and fire). Different rates were used for pre- and post-1800 since this was when grazing began to increase with a delayed regime shift occurring in the ca.1950s. Input variables and their associated assumption are described in Table 7.2 and Table 7.3. Foundational literature reviewed and considered in the model were related to resilience theory (including alternative stable states and management thresholds), patch-mosaic landscapes, and disturbance and the competitive exclusion principle (the success to the successful systems archetype) (Biggs and Rogers, 2003; Bond, 2019; Gillson et al., 2019; Kim and Anderson, 1998; Milton, 2007; Radloff et al., 2014; Wang and Liu, 2020; Weiner et al., 2019).

### 7.2.4 Model validation testing, debugging and verification

Model validation testing is carried out throughout all five phases of the SD modelling process to increase the confidence in the model structure and behaviour, and therefore provides an opportunity to improve the model iteratively (Figure 4.4). In terms of Validation Hierarchy and Model Hierarchy (Groesser and Schwaninger, 2012; Schwaninger and Groesser, 2009), the suite of model validation tests used in this study (Table 4.6) covers types/categories that have a good spread of varying degrees of complexity (see first three columns of Table 7.4). A summary of the supporting documentation for the behaviour sensitivity results are shown in Appendix Figure 10.25 and Table 10.10, and the unabridged sensitivity analysis results will be made available as supporting material upon peer-review publication. Considerations as to when model validation was discontinued is also described below.

***Table 7.4: Summary of model validation results. Varying degrees of complexity are associated with the suite of validation results. Domains of Validation (model structure, behaviour, context); Levels of Resolution (micro, meso, macro); and Levels of Complexity (I-V) (Groesser and Schwaninger, 2012; Schwaninger and Groesser, 2009).***

| Domains of validation; level of resolution; and level of complexity | Validation hierarchy | Model hierarchy | Validation results |
|---|---|---|---|
| **Model structures tested on micro level (I and II)** | **Elementary level** | **Elements and combinations thereof** | Before and during model development, structural and parameter confirmation tests were carried out, model equations were inspected with no unit errors. Furthermore, multi-stakeholder social learning workshop (4 July 2019) allowed for expert validation to refine the model's accuracy by confirming the structure, capturing mental models and setting the model boundary. Modelled results responses to direct extreme values of each flow equation as expected. |
| | **Simple dynamics** | **Feedback loop** | Extreme-conditions tests were conducted on the variables with the lowest extreme = 0 and the highest extreme =1 and the model responded as expected: Effect of fire on decrease in Asteraceous shrubs; Effect of fire on decrease in Renosterbos; Effect of grazing on decreasing Asteraceous shrubs; Effect of Renosterbos homogenisation on fire. An integration error test was performed, and the model is not sensitive to a change from Eula to RK2 or RK4, therefore there are no data probability issues. |
| **Model behaviour to test large meso structures (III)** | **Multiple dynamics** | **Combination of feedback loops** | Behaviour sensitivity was conducted on 18 variables (Appendix Figure 10.25 and Table 10.10) of which the five most sensitive were: Asteraceous pollen relative decrease rate; Post-1800s fire relative increase rate; Post-1800s fire relative decrease rate; Standard effect of grazing; and Standard effect of fire on Asteraceae. Based on the unprecedented increase in Renosterbos pollen since ca. 1950s, the effect of Renosterbos homogenisation on increasing fire was expected to be highly sensitive but it showed low sensitivity. Further policy sensitivity analysis is recommended to identify more suitable leverage points for policy. Manual clearing of invasive Renosterbos is expensive and managers would need incentives to manage grazing and fire to maintain current levels of plant biodiversity. |
| **Model behaviour to test macro structures (IV)** | **Full dynamics** | **Complete model** | Stakeholders provided context via values-based interpretations, which included their perceptions and expert opinion of the land management system and drivers of change. Expert input was captured in a systems map (mental models as tacit knowledge, Appendix Figure 10.23) and used to validate the current dynamic hypothesis (Figure 7.5). The simulation model reproduced the trends far back in the past (ca.1300 years ago) and showed an increase in fire and grazing (due to livestock and reintroduced large indigenous herbivores) and a decrease in local plant biodiversity since ca. 1950s. However, the modelled data differs from the historical data point-by-point thus not showing the short-term oscillations but the general patterns. |
| **Model context to test micro, meso and macro structures (V)** | **Meta level** | **Context of model** | Model purpose was articulated once the broader sustainable land-use management problems were defined during the multi-stakeholder social-learning workshop. Elandsberg PNR was chosen as a case study and the model purpose clearly stated: *To assist reserve managers set land-use guidelines/strategies on future grazing and fire that would protect local plant biodiversity and therefore ecosystem function and resilience.* |

#### *7.2.4.1 Considerations around the Validation Cessation Threshold (VCT)*

Despite some factors resulting in a relatively higher VCT, ceasing efforts for formal testing was due to factors such as SD training and upskilling of the modeller, access to rare site-specific data from the Elandsberg PNR palaeo-record and that this chapter is a proof of concept to meet Objective 4 and Component B and C. Therefore, the modelling component of this study was not the worst-case modelling situation and validation cessation (Appendix Figure 10.1) was reached as explained below (Groesser and Schwaninger, 2012).

- **Higher data intensity**: Although the literature shows a reasonable body of ecological research of fire and grazing effects on plant biodiversity within the CFR, the understanding of long-term variability within the Middle Berg River Catchment remains limited (Figure 2.2). We address these gaps in knowledge by using the multi-proxy, high temporal resolution palaeoecological dataset from Elandsberg PNR case study. Together with selected palaeo-proxy data (fossil pollen, macro-charcoal and coprophilous fungal spores - Figure 7.7), insights from historical grazing data over the period 1928-1994 recorded in the Agricultural Censuses (Statistics South Africa, 2010) and 1994-2019 recorded in Elandsberg Game Management Plan were used for model development (see Appendix Table 10.8 and Figure 7.6).
- **Moderate level of modeller's expertise**: Given that this is an interdisciplinary PhD project, my relatively low-level expertise in SD modelling was a contributing factor to the higher VCT. However, basic SD training and preparation increased my level of modelling expertise to sufficiently conduct the PSD component of the research Appendix section 10.4.1).
- **Low relative importance/risk of decision:** A large contributing factor that also lowered the VCT is that this case study is used as a proof-of-concept test whether it is feasible to use long-term palaeoecological data in an SD model. The end-product is of low decision risk since it is designed for management decision-support as opposed to decision-making.
- **Low level of target group's experience with modelling:** Although all stakeholders that participated in this project are experts in land-use management and governance within the region, they have a low level of SD experience. Additionally, most stakeholders had not considered using either of the disciplines (palaeoecology and system dynamics) to inform their work prior to project inception in 2016.
- **Low level of target group's expectations:** All stakeholders were aware that their contribution was a part of an applied palaeoecological research study and SD model development was exploratory in nature, thus their expectations were intentionally low. At the time of early model

development (phase 1 and 2 of Figure 4.4 and Table 4.5) the model had been designed for policymakers and practitioners in a multi-sector target group and forecasting their expectations was not possible. However, future PSD modelling (see conceptual model in Figure 7.1) would benefit from participation of a broader target group located near all three study sites, including, conservation practitioners, commercial farmers, local communities, key land management institutions from local and regional government, as well as a scientific audience interested in SES research, particularly conservation researchers. Elandsberg PNR reserve managers were chosen as the target group, thus decreasing the time and financial costs of model verification and validation. The SFD considers their perception on the loss of plant biodiversity in terms of land-use management at Elandsberg PNR.

## 7.3 Simulation model results: Model behaviour of preliminary scenario and policy analysis of the Ecological Model

The policy space for biodiversity conservation at Elandsberg PNR is defined in Table 7.5 and Table 7.6. The Ecological Model reproduces threshold behaviour that could be described as a tipping point, crossing of an ecological threshold or a regime shift or transition to an alternative stable state (Folke et al., 2004; Holling, 1973; Ludwig et al., 1997; Walker and Meyers, 2004). This behaviour is likely due to the shift in loop dominance from Asteraceous shrubs dominance (R1) to unpalatable Renosterbos dominance (R2) (Figure 7.9) driven by increased grazing pressure by sheep and cattle, which began in the early ca. AD 1800s and eventually caused a regime shift during agricultural intensification in ca. AD 1950s. The dominance of Renosterbos was exacerbated by even higher grazing levels when large indigenous herbivores (LIHs) were reintroduced following Elandsberg PNR's proclamation in AD 1973 (Figure 7.7 and
Figure 7.13d). Reintroduced LIHs included: eland, blue wildebeest, black wildebeest, zebra, red hartebees, gemsbok, bontebok and springbok springbok (Wooding pers. comm. 2012) (Appendix Table 10.8). Furthermore, an increase in fire levels since ca. AD 1950s (
Figure 7.13c), likely linked to late 20$^{th}$ century warming and increased human population, reinforced the Renosterbos dominance. Competitive exclusion by unpalatable Renosterbos (loop R2), caused fire persistence (loop R3) and influences grazing dynamics (loop B1) (Figure 7.9), and thus a loss of plant biodiversity in the landscape.

Developing future scenarios are an important tool for policy analysts in government and industry and is an important contribution in the present study as well (Becker, 1983; Roura-Pascual et al., 2021). A

snapshot of plant biodiversity and fire levels in the future (AD 2050 and AD 2100) are shown in Table 7.6, whereas Figure 7.14 and Figure 7.15 show how trends in vegetation dynamics evolve from AD 2020-2100. Noteworthy results for each modelled scenario are highlighted in Table 7.6. The Base case scenario represents business as usual if the rates of changes in the abundance of Asteraceous shrubs, Renosterbos, fire and grazing levels stay constant over time (
Figure 7.13a-d). As expected, there is a decrease in plant biodiversity (increase in Renosterbos and decrease in Asteraceous shrubs) (Figure 7.14). Blue shading in Table 7.6 shows a more suitable ratio (0.35 in 2050 and 0.19 in 2100) if managers only control grazing (Policy 2, Table 7.5) and do not invest in adequate fire prevention and control, Asteraceae pollen would only reach 7.68% by 2100 and the macro-charcoal levels would be 1.07 times higher (1875 particles cm-3) than the Base case (1760 particles cm-3).

At the multi-stakeholder engagement workshop, stakeholders expressed their concerns about current levels of fire - "*Fire is part of the ecosystem, it is required for propagation, but the fire frequency is too high*" and *"...fire is an interesting problem as it drains the economy – and people sue each other*". Stakeholders mentioned that they are already taking fire prevention precautions by not using certain agricultural machines on high Fire Danger Index (FDI) days, but that more prevention and control measures supported by systemic structures (e.g. fire protection services) are needed for to help raise awareness, increase collaboration with fire protection services and lobbying the authorities for additional financial resources for fire control. Climate was not modelled using the Intergovernmental Panel on Climate Change (IPCC) global circulation models and the ARs due to the relatively small size of the model used as a ‘proof of concept’ in the present study, and thus the modelling decision that ‘climate’ was an excluded as a variable but the increase in fire was the proxy for climate impacting land-use management. However, using the official climate predictions from the IPCC would be an interesting inclusion during future expansion of scenario analysis to improve upon the model prototype. Given the boundary decision in this model, the effects of a hot dry climate (Scenario 2) and drought (Scenario 1) impacted fire by increasing the FDI and therefore increased the fire rate, however the ratio of Asteraceae to Renosterbos pollen (0.19 in 2050 and 0.08 in 2100 - Table 7.6) did not change significantly compared to the base case – see pink shading in Table 7.6.

If managers decrease fire levels only, there will be some recovery in Asteraceous shrubs but unpalatable Renosterbos still persists (Figure 7.14) Simulated fire control (Policy 1.1) resulted in less local fires (macro-charcoal 1248 particles/$cm^3$) compared to fire prevention (Policy 1.2) (macro-charcoal 1732 particles/cm$^3$), however fire prevention is not beneficial for increasing biodiversity (ratio 0.19 in AD 2050 and 0.09 in AD 2100 - Table 7.6) since Renosterveld and Fynbos vegetation is fire-prone and requires fire for regeneration.

***Table 7.5: Summary of parameters that define the policy space for land-use management at Elandsberg Private Nature Reserve***

| | **Base** | **Scenario 1: *10 year Cape drought*** | **Scenario 2: *Climate warming*** |
|---|---|---|---|
| **Base** | Initial conditions for fire and grazing levels. Management interventions are reactive rather than proactive and climate change adaptation is not a priority. | The current Cape drought is ongoing in various regions since 2015. This is the worst drought in 100 years. Assumption that dry conditions will persist for another 6 years before receiving improved average rainfall. Improved data needs to be obtained for this region to make this more realistic since the size of change is an assumed rate (0.1). Additional data from Fynbos fire dynamics experts are required. | Scenario 2 assumes that fires will increase at a certain steady rate after the year 2020. The increase in fire increase rate (0.001 slope) is an assumption about how climate change increases the Fire Danger Index days and there are more fires due to hot and dry conditions (Stakeholder Workshop 1- July 2019) and van Wilgen et al. (2012). |
| **Policy 1.1: *Fire prevention*** | Dcreases the fire increase rate (by 0.1 from 2020) onwards due to increased fire prevention in the form of public awareness raising about high Fire Danger Index days. | Scenario 1 - increase fire increase rate (0.23 + 0.1) for 10 years.<br><br>Decrease fire increase rate by 0.1 from 2020 onwards. | Scenario 2 - increase fire increase rate by 0.23 + 0.001 slope.<br><br>Decrease fire increase rate by 0.1 from 2020 onwards. |
| **Policy 1.2: *Fire control*** | Increase fire decrease rate by enforcing fire break laws and improving Fire Support Association initiatives such as training during controlled burns Assumption: 0.1 rate to decrease fire per year. | Scenario 1 - increase fire increase rate (0.23 + 0.1) for 10 years.<br><br>Increase fire decrease rate by 0.1 from 2020 onwards. | Scenario 2 - increase fire increase rate by 0.23 + 0.001 slope.<br><br>Increase fire decrease rate by 0.1 from 2020 onwards. |
| **Policy 2: *Decrease to 9.5% grazing only*** | Policy 2 is the decision taken by Elandsberg PNR to reduce their stocking number to the pre-1950s level (9.35% therefore 0.095) (Forbes et al. 2018) for certain period of time (5, 10 or 50 years). | Scenario 1 - increase fire increase rate (0.23 + 0.1) for 10 years.<br><br>Grazing levels decreased to 0.095. | Scenario 2 - increase fire increase rate by 0.23 + 0.001 slope.<br><br>Grazing levels decreased to 0.095. |
| **Policy 3: *Adaptive grazing-fire management*** | Managers use adaptive grazing-fire management to decrease grazing to the average pre-1950s levels (9.35%) for a certain period of time (5, 10 or 50 years) and then also implement management that decreases fire (via fire prevention - Policy 1.1 and control - Policy 1.2). Grazing levels decreased to 0.095 for 50 years then return to 0.53 + decrease fire increase rate by 0.1 + increase fire decrease rate by 0.1. | Scenario 1 - increase fire increase rate (0.23 + 0.1) for 10 years.<br><br>Grazing levels decreased to 0.095 for 50 years then return to 0.53 + decrease fire increase rate by 0.1 + increase fire decrease rate by 0.1. | Scenario 2 - increase fire increase rate by 0.23 + 0.001 slope.<br><br>Grazing levels decreased to 0.095 for 50 years then return to 0.53 + decrease fire increase rate by 0.1 + increase fire decrease rate by 0.1. |

Interestingly, the decision point variable "*Duration of policy: Time to adjust grazing management*" related to the Adaptive grazing-fire management (Policy 3, Table 7.5) has varied consequences depending on the number of years. Considering the priorities of the current stakeholders managing Elandsberg PNR, it may be unfavourable for a thriving private nature reserve to decrease its large indigenous herbivore stocking rates for 50 years, thus the duration could be decreased to 10 or 5 years. When only implementing the policy for 5-10 years before adjusting grazing back to its current levels, Renosterbos pollen abundance at 2100 would only be slightly higher (55%) than if adjusted after 50 years (53%). Similarly, Asteraceous shrubs would be between 10.6-10.97% after adaptive management for 5-10 years, which is not much less than 11.88% after 50 years. Furthermore, the ratio of Asteraceous to Renosterbos pollen at 2050 and 2100 under various scenarios is shown in the last two columns of Table 7.6. Before the regime shift occurred around the ca. AD 1950s, Asteraceous shrubs and Renosterbos were in equilibrium (i.e. Regime V-1 with no abrupt changes in ecosystem services and drivers) for at least 1200 years (Forbes et al., 2018), with Asteraceous shrubs dominant in the system (see R1 of Figure 7.9). This equilibrium is expressed in terms of a ratio or fraction. On average the pre-1950s ratio of Asteraceous to Renosterbos pollen was 2.1 whereas the post-1950s ratio was low, 0.4 (Forbes et al., 2018). Ideally policy and management would aim to return levels of grazing, fire and plant biodiversity to that of a pre-1950s baseline (i.e. a ratio of 2.1). However, simulation results indicate that neither reversing this shift in abundance to pre-1950s levels nor maintaining the post-1950s levels may be possible in the future. Instead Renosterbos would continue increasing with time. Policy 3 may be the most reasonable intervention to maintain this new current system state by not crossing a land cover change management threshold that is calibrated to a Asteraceae:Renosterbos pollen ratio of 0.22 - 0.35 (see green shading in Table 7.6). This will decrease Renosterbos slightly as well as control the high fire frequency problem as well.

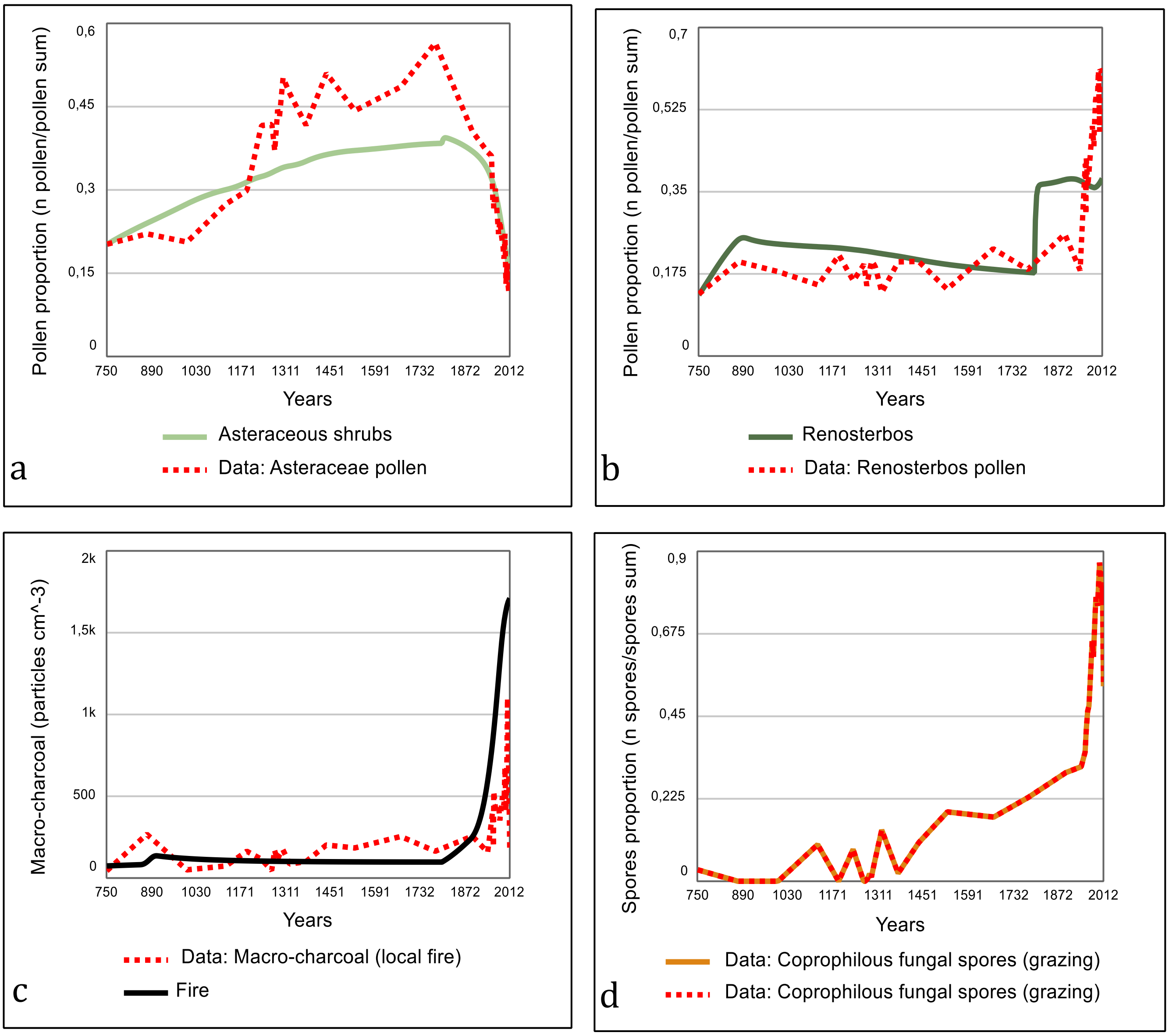



***Figure 7.13: Graph showing base run behaviour compared to the reference modes (red dotted line) of the three key stocks: (a) Asteraceous shrubs (Asteraceae pollen), (b) Renosterbos (Renosterbos pollen) and (c) Fire (macro-charcoal). (d) Coprophilous fungal spore data was exogenous in this model.***

***Table 7.6: Summary of results from the scenario and policy analyses for the Ecological Model[1]. Ratio of Asteraceae to Renosterbos pollen calculated as per the modelled output. Noteworthy results are highlighted and described in the text.***

| | Decision points and initial values since 2021 | | | Palatable Asteraceae (%) | | Unpalatable Renosterbos (%) | | Macro-charcoal (particles/cm$^3$) | | Ratio of Asteraceae to Renosterbos | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Scenario name** | **Fire increase rate** | **Fire decrease rate** | **Grazing level** | **2050** | **2100** | **2050** | **2100** | **2050** | **2100** | **2050** | **2100** |
| **Base case –1–** | 0.23 | 0.05 | 0.53 | 8.17 | 3.71 | 41.73 | 44.45 | 1754 | 1760 | 0.20 | 0.08 |
| **10 year Cape drought conditions (Scenario 1)** | 0.23 + 0.1 | 0.05 | 0.53 | 7.94 | 3.61 | 41.85 | 44.51 | 1755 | 1760 | 0.19 | 0.08 |
| **Climate warming (Scenario 2)** | 0.23 + 0.001 slope | 0.05 | 0.53 | 8.02 | 3.36 | 41.06 | 42.96 | 1811 | 1879 | 0.20 | 0.08 |
| **General climate change –2–** | 0.23 + 0.1 + 0.001 slope | 0.05 | 0.53 | 7.80 | 3.27 | 41.18 | 43.01 | 1812 | 1879 | 0.19 | 0.08 |
| **Fire control –3–** | 0.23 + 0.1 + 0.001 slope | 0.05 + 0.1 | 0.53 | 10.32 | 6.97 | 50.20 | 49.91 | 1058 | 1248 | 0.21 | 0.14 |
| **Fire prevention –4–** | (0.23 + 0.1 + 0.001 slope) - 0.1 | 0.05 | 0.53 | 8.56 | 4.21 | 44.04 | 44.72 | 1571 | 1723 | 0.19 | 0.09 |
| **Decrease to 9.5% grazing only –5–** | 0.23 + 0.1 + 0.001 slope | 0.05 | 0.095 | 13.31 | 7.68 | 38.23 | 40.39 | 1799 | 1875 | **0.35** | **0.19** |
| **5 yrs Adaptive management –2–** | (0.23 + 0.1 + 0.001 slope) until after 5 years then -0.1 | 0.05 until after 5 years then + 0.1 | 0.095 for 5 years then return to 0.53 | 12.26 | **10.60** | 56.37 | 55.37 | 513 | **722** | 0.22 | 0.19 |
| **10 yrs Adaptive management –3–** | (0.23 + 0.1 + 0.001 slope) until after 10 years then -0.1 | 0.05 until after 10 years then + 0.1 | 0.095 for 10 years then return to 0.53 | 12.71 | 10.97 | 55.75 | 55.07 | 524 | 721 | 0.23 | 0.20 |
| **50 yrs Adaptive management –4–** | (0.23 + 0.1 + 0.001 slope) until after 50 years then -0.1 | 0.05 until after 50 years then + 0.1 | 0.095 for 50 years then return to 0.53 | 13.31 | **11.88** | 38.23 | 53.30 | 1799 | **770** | **0.35** | **0.22** |

[1] Link to the Stella Architect model files for the Ecological Model: https://drive.google.com/file/d/1VrBckYm3dsFte7Vy_PNpnxUFRyKV4j4T/view?usp=sharing. Upon publication the model will be hosted on isee Exchange and accessible online for free.

## 7.4 Discussion: Policy strategy and Management implications

### 7.4.1 Insights for ecological restoration

Based on simulation results from the Ecological Model, the following is recommended for biodiversity conservation and ecotourism at Elandsberg PNR: Reduction of land-use disturbance by implementing appropriate adaptive grazing-fire management practices that decrease grazing pressure and the rate of fire (Figure 7.15) to maintain or improve current levels of plant biodiversity and prevent a future regime shift to an alternative stable state, Degraded Renosterveld. Palaeoecological results indicates that although increased grazing since the ca. AD 1800s (during a period of European Settler Agriculture) may have gradually triggered a tipping point, the effects were only evident by the ca. AD 1950s during agricultural intensification when there was a shift in dominance from Asteraceous shrubs to unpalatable Renosterbos and therefore a loss in plant biodiversity. However, as evident from fire history maps of Elandsberg PNR, varying post fire ages create a patchy/heterogeneous landscape with a combination of different vegetation units (Figure 3.5). Thus, together with an increase in Renosterbos, there are still patches of Asteraceous shrubs, grasses and thicket (that act as fire refugia), which contribute to the current maintenance of plant biodiversity. System dynamics helps us go a step further by thinking about long-term change in the future and what appropriate management interventions can maintain heterogeneity and the current regime where the ratio of Asteraceae:Renosterbos pollen is between 0.22 and 0.35 (Table 7.6).

From palaeoecological analysis alone, Forbes et al. (2018) speculated that the presence of these important vegetation units indicates that the current slightly degraded state (Regime V-2: Renosterbos dominance from 1950s-present) is likely to be reversible if reserve managers take immediate action. On the contrary, future scenarios analysis conducted via the current Ecological Model highlighted that reversing this regime shift may not be as simple as reducing grazing and fire frequency. Thus, simulation results suggest hysteresis (Beisner et al., 2008; Ludwig et al., 1997) (Beisner et al., 2008; Dearing et al., 2015; González Sagrario et al., 2020; Jackson and Wood, 2018; Ludwig et al., 1997) of the post-1950s alternative stable state (Regime V-2). The new ecological regime dominated by unpalatable Renosterbos together with a higher fire frequency is less desirable, and probably more “negatively resilient” in its current state as it shows resistance (Grace and Pope, 2015; Lake, 2013; Oliver et al., 2015), and thus difficulty in reversing or changing the system back to how it was (Walker and Meyers, 2004). If reserve managers intervene with a quick fix in the 5 yr Adaptive Management scenario, Asteraceous shrubs abundance could increase slightly to 10.6% by 2100 compared to the

Base case scenario (3.71%) (Figure 7.15 and Table 7.6), but the abundance of undesirable Renosterbos would also increase (44.45% for the Base case scenario to 53.37% in the 5 yr Adaptive Management scenario).

The hysteresis of this current stable state is likely due to the starting conditions of the landscape which included high unprecedented levels of Renosterbos, grazing and fire when the area was declared a nature reserve in AD 1973. It is essential that restoration ecologists consider the starting conditions of ecosystem services and disturbance indicators such as fire, herbivory and invasive alien plants (IAPs) when aiming to set conservation targets and implement restoration activities. In other words, based on past variability seen in the 1300-year-old VANG record, ecological character (Gell, 2012, 2017) has shifted from a pre-1950s regime to a post-1950s regime and the use of AD 1973 as a reference state in Elandsberg PNR landscape is unrealistic for the setting of biodiversity conservation targets as the landscape was already degraded. As agricultural activities in Elandsberg PNR ceased after the proclamation of the reserve, the ecosystem at the time was likely considered to be functioning naturally and the management decision to reintroduce LIHs was justified on the grounds that such herbivores had been present before being replaced by domestic livestock.

If we apply the Iceberg Model then the observed undesirable trend of an increase in unpalatable Renosterbos is due to deeper systemic structures of connected, interacting patterns. These include overgrazing and an increase in the incidence of fire which in turn lead to a decrease in plant biodiversity. However, even deeper mental models underpin these patterns, for example, when inappropriate land-use management baselines shifted as land-use priorities changed from agriculture to conservation in ca. AD 1970s (Forbes et al., 2018). The beliefs, values and assumptions considered the reference state/starting conditions in AD 1973 as the "natural ecological character". Therefore, high fire frequency and high levels of herbivory from reintroduced LIHs were considered defensible baselines for land-use management. Together, the palaeoecological reference modes (Figure 7.7) and qualitative CLD (Figure 7.9) suggest that the delay between the effects that grazing had on Renosterbos homogenisation increased gradually from ca. AD 1800s until ca. AD 1950 when an ecological threshold was crossed, after which the new dominant feedbacks maintained the new less desirable state (Figure 2.6).

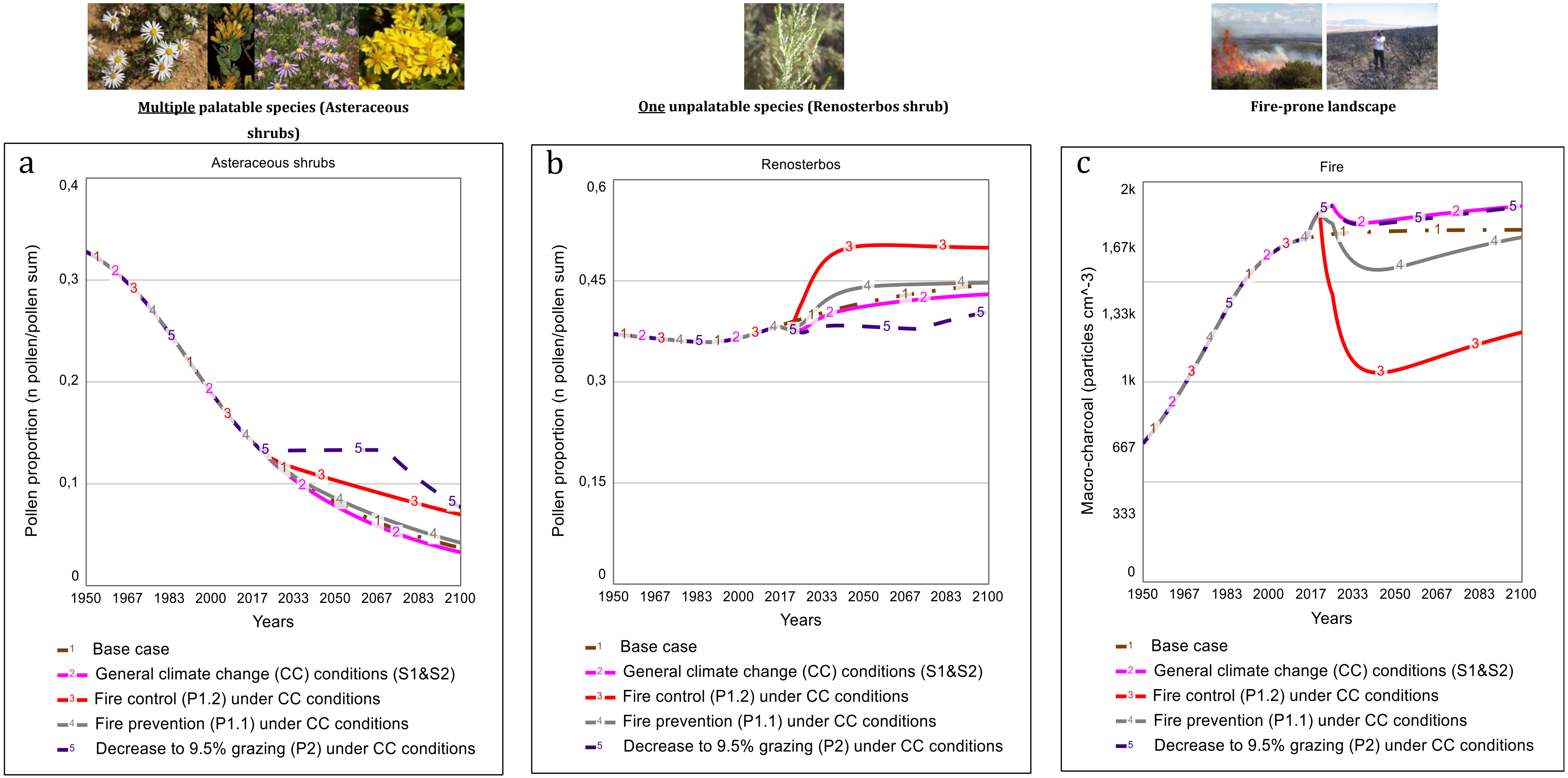
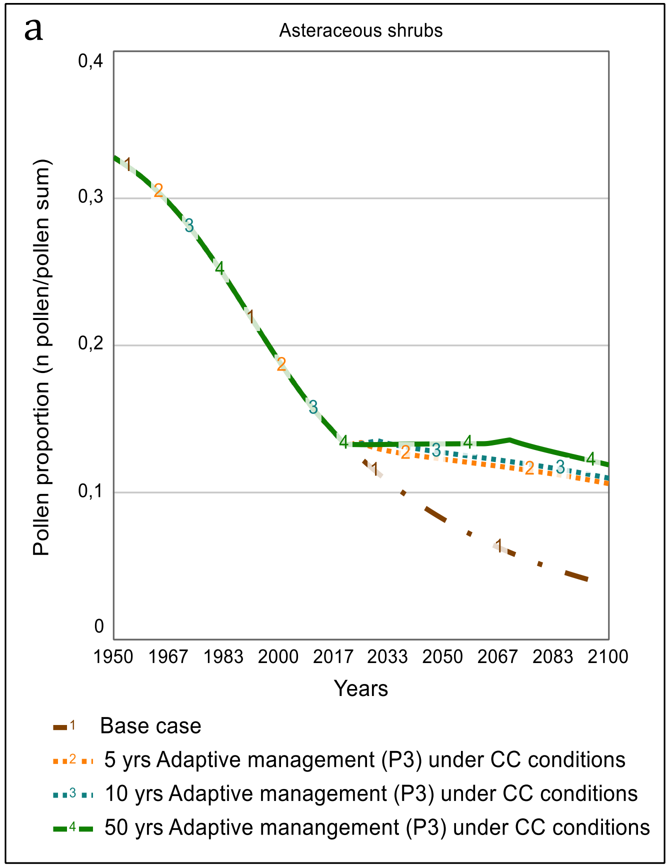


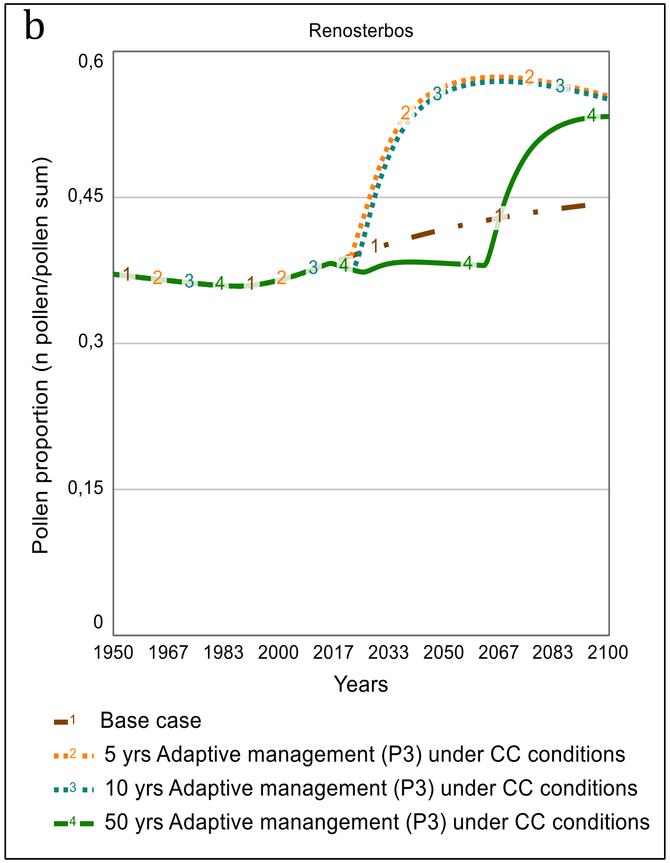

***Figure 7.14: Five scenario analysis outputs for the Ecological Model showing changes in plant biodiversity (pollen proportion data of (a) multiple Asteraceae species and (b) one unpalatable species, Renosterbos) and (c) fire (macro-charcoal) over time (1950-2100) for the Elandsberg PNR.***

**Multiple palatable species (Asteraceous shrubs)**

**One unpalatable species (Renosterbos shrub)**

**Fire-prone landscape**



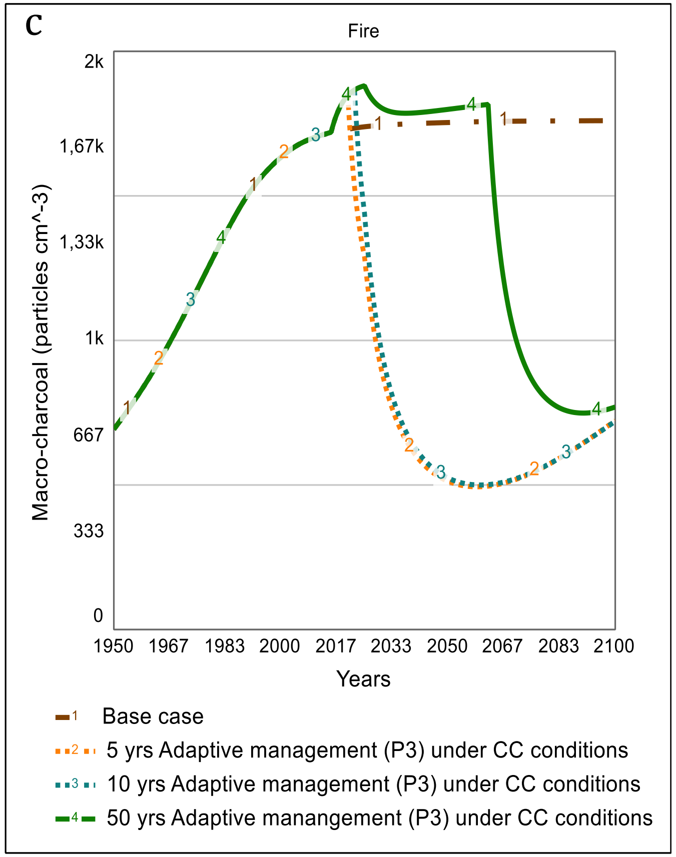




***Figure 7.15: Expansion of Policy 3's Adaptive grazing-fire management analysis outputs (5, 10 and 50 years) compared to the Base case for the Ecological Model showing changes in plant biodiversity (pollen proportion data of (a) multiple Asteraceae species and (b) one unpalatable species, Renosterbos) and (c) fire (macro-charcoal) over time (1950-2100) for the Elandsberg PNR.***

Even if managers intervene by decreasing grazing and fire to pre-1950s levels, the results suggest that it would be too little to achieve the decision-makers goal of reducing the abundance of Renosterbos in the landscape (Figure 7.15). However, the sensitive leverage points were not intuitive without combining the long-term palaeo-perspective with future scenarios analysis (Process Steps 8-12, Table 4.5). Furthermore, the decision to reintroduce LIHs without knowledge of past and present acting disturbances and the ecological character of the previous intact vegetation condition is also a sensitive leverage point for change (Meadows, 2008), or in this case, resistance to change.

Admittedly, it may prove difficult for stakeholders to implement the recommendations above, especially those related to decreasing the large indigenous herbivore stock since this will directly influence revenue generated by ecotourism activities at Elandsberg PNR. However, Elandsberg PNR currently manages the stock according to a maximum load capacity (1100 LSU)) based on current reserve size and rainfall. As of 2012 their stock capacity was four times smaller (LSU 251) than the recommended maximum (Wooding pers. comm. 2014). The palaeoecological data also shows a slight decline in fire and grazing since ca. AD 2008, which confirms the efforts of managers reducing the stock capacity. However, alarming trends of decreasing Asteraceous shrubs and increasing Renosterbos persists (Forbes et al., 2018). Given the evidence for possible hysteresis in the future, we recommend that managers maintain and monitor current land-use practices or restore ecological character to avoid degradation of the ecosystem leading to a hysteretic reorganisation once critical ecological thresholds are crossed. Reversibility of an even more Degraded Renosterveld state in the future (Forbes et al., 2018; Slingsby et al., 2014) is highly unlikely.

Therefore, implications of the results could justify active restoration of previous vegetation composition including clearing of Renosterbos to rehabilitate these degraded areas. However, a previous study revealed that the cost of intensive manual clearing of Renosterbos can be high and that clearing techniques are challenging (Millar et al., 2007; Palmer, 2010). Elandsberg PNR has undergone different types of land-uses for many centuries before its proclamation as a conservation area (Becker, 1996). It is therefore an example of a SES that must be managed for several ecosystem services (provisioning, supporting, regulating and cultural services). It is important to consider the effects of the regime shift on ecological, social and economic well-being, for example the tension between wildlife tourism and other ecosystem services such as plant biodiversity, and resilience to climate change. Despite the decline in plant biodiversity, reintroduced LIHs are beneficial for recreation and ecotourism (i.e., cultural services). Therefore, the gain in cultural services since the ca. AD 1970s impacts human well-being positively since increased number of tourists would increase the per capita income of the employees at the reserve and economic growth in the region (Midgley et al., 2014) (Table 1.2).

However, if Renosterbos, herbivory and fire increase further in the future, this very same human well-being indicator (ecotourism representing economic condition) could decline. Land degradation would eventually drive the system to no longer support the ecological carrying capacity (ECC) of LIHs (Figure 7.6). If another regime shift had to occur in the future, the landscape would become homogenised with increased unpalatable Renosterbos and the loss of other vegetation units (e.g. Asteraceous shrubs, thicket and indigenous grasses) resulting in a loss in the utilitarian value of biodiversity (provisioning and supporting service), as well as a decrease in soil erosion regulation (regulating service) since there would be more bare ground and increased runoff. Such future scenarios would not meet long-term sustainability goals for this conservation site since it is inevitable that biodiversity and associated ecotourism (i.e. number of tourists visiting Elandsberg PNR to view wild game and plant biodiversity) could decline impacting income generated and therefore decreasing human well-being.

### 7.4.2 Land Management Decision-Support Tool

When blending methods from a variety of disciplines, it is important to foreground the politics and social dynamics related to not only how knowledge is produced but also how it is taken up outside of academia (Biermann et al., 2020; Roux et al., 2017). Thus, the applied palaeoecological Community of Practice (CoP) needs to internalise the responsibility of sharing palaeo-outputs that are translated and packaged into usable formats for all relevant stakeholder groups (whether they are from the academic, public, private or civil society sector). Presented with Storylines and Scenarios, and with an interactive modelling interface, stakeholders can engage with the findings in a way that encourages reflection (Taylor et al., 2016) and co-learning to provide insights for action such as planning preventative measures at various timescales and levels of low to high impact. Such long-term data helps ground potential future scenarios in the realm of what is ecologically possible (Jackson et al., 2009; Rice et al., 2009), for example, stakeholders are more willing to consider multi-decadal droughts, landscape-altering fires, and rapid Renosterbos or IAPs species invasions as realistic possibilities for the future since palaeoecological evidence shows that they have happened in the past (section 2.2.3). Furthermore, system dynamics modelling allows stakeholders to collectively envision a desired land management future and consider alternative future scenarios (section 2.4.2.2).

The Land Management Decision-Support Tool as illustrated by the screenshot in Figure 7.16 and is accessible online for free (https://exchange.iseesystems.com/public/cherie-dirk/ecological-model-land-management-decision-support-tool) in its current form can now be regarded as explicit knowledge since the tacit knowledge from stakeholders is captured in the model (section 7.2.3). It can be used as a boundary object to facilitate discussion about any surprising simulation results and explore future scenarios for implementation (section 2.5.3). This allows for the application of the end-product, including, education and outreach regarding plant biodiversity, grazing and fire management (Tidwell et al., 2004; Winz et al., 2009). With this interactive modelling interface, multiple stakeholders can engage with the system dynamics model findings and learn about the value of the long-term information presented to them (Fischer and Riechers, 2019; Star and Griesemer, 1989). When stakeholders recognise the value of the information presented to them, they can begin to engage with the findings in a tangible way that may encourage them to plan for preventative measures at various timescales and levels of low to high impact (Armstrong McKay et al., 2019; Hossain et al., 2020; Jackson and Hobbs, 2009; Menendez et al., 2020; Rice et al., 2009). Moreover, as the PSD modelling process is iterative (Figure 4.4), use of the Land Management Decision-Support Tool will improve the model prototype by generating dialogue to further articulate the dynamic problem and thus be another form of model validation.

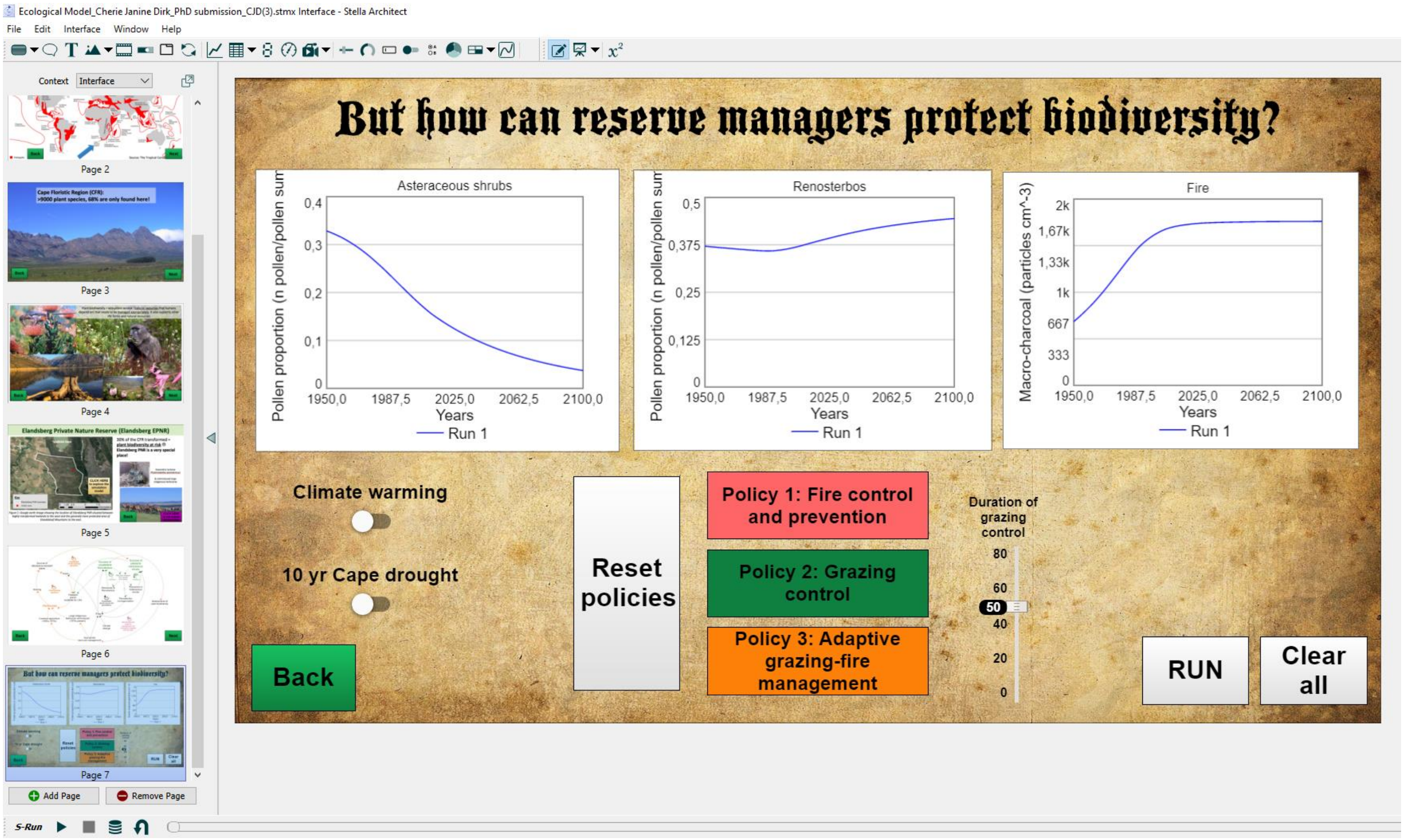



*Figure 7.16: Screenshot of the Ecological Model Stella user-interface to be used as a Land Management Decision-Support Tool. Available via open access at the following link: https://exchange.iseesystems.com/public/cherie-dirk/ecological-model-land-management-decision-support-tool.*

### 7.4.3 Recommendations for CFR's climate change and biodiversity policy context

Findings from the present study can inform ecosystem resilience assessment policy. A long-term perspective which includes a past-present-future lens of environmental change, provides improved evidence of the variability of environmental change. It also provides a unique opportunity to identify drivers of change such as climate change and land-use disturbance as a result of agriculture or conservation. This facilitates a strengthened understanding of which factors drive behaviour over time so that management can be improved. High-resolution, multi-proxy palaeoecological studies combined with PSD facilitates researchers to cross boundaries between neoecological and palaeoecological research, as well as inform conservation practice. Thus, there is potential to include such findings on the variability of ecosystem services in regional ecosystem assessments, with relevance for global change assessments (e.g. Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Brondizio et al., 2019; Díaz et al., 2015; Jeffers et al., 2015)) and the understanding of resilience in agro-ecological landscapes (Armstrong McKay et al., 2019; Cavender-Bares et al., 2015; Dearing et al., 2012).

This study shows that participatory approaches, dynamic- and causal-thinking that assists in collectively understanding complex SESs to develop insights for sustainable ecosystem management is possible. As decision-making was not the objective of the present study but rather the aim to provide management decision-support (Component C, Figure 1.2), I contend that bridging the research-implementation gap, or the knowing-doing gap (Knight et al. 2008), for mainstreaming applied palaeoecology requires a two-step process: (1) Development of strategic action plan to include palaeoecological insights into biodiversity and land-use planning policy and implementation; and (2) leveraging existing networks (e.g.) and polycentric governance to mainstream applied palaeoecology into non-biodiversity sectors such as production sectors. Such can be achieved through fostering and supporting social learning among farmers (Rouget et al., 2014; Wheeler et al., 2019) through extension services and other face-to-face farmer support (e.g. Agtergroenberg Conservancy, Overberg Renosterveld Conservation Trust) and conservation stewardship agreements (see sections 3.2.3, 3.4.1 and 3.4.2).

Based on insights from the present study, we recommend that policy should incorporate the following elements: (1) Education, training and awareness amongst public and private reserve managers (and interested parties) on the importance of landscape history and a past-present-future lens of environmental change. Baseline assessments to evaluate past and current levels of ecosystem services to establish the natural ecological character and determine whether the current starting conditions (current ecological character) are an ideal reference. Stakeholders should be encouraged to share about their perceptions of past and future change and various conservation practices that they and their collaborators (including government) could invest in which would maintain ecosystem integrity and still allow for a thriving ecotourism sector. Those working with long-term data should engage with these stakeholders and find ways of using and presenting the data that are accessible and useful. (2) Mainstreaming the relevance of plant biodiversity and appropriate fire levels so that it is not only considered in the environmental/conservation sector but also in agriculture, urban development, and general civil society. Current fire levels have been described as worrying by several stakeholders (farmers, conservation practitioners, and government officials), and improved policies and implementation are needed to tackle unprecedented levels of fire, especially given the effects of climate change. (3) To operationalise systemic resilience (Helfgott, 2018) and converge multiple knowledge systems (Oteros-Rozas et al., 2015), novel participatory processes (including GMB, social learning and future scenario planning techniques) are needed. Through the interrogation of palaeo-data and future policy and scenario analysis, stakeholders can collectively reflect on long-term change and envision different future scenarios that may be outside of their living memory. Participatory scenario planning techniques evoke stakeholders’ ambitions, plans and perceptions of change (Wollenberg et al., 2000) and stimulate dialogue on adaptation measures to achieve a shared vision of the future. Thus, contextual planning and management can be conducted at a local/landscape and catchment scale to strive for a 'desirable scenario' which maintains biodiversity and socio-economic benefits for conservation.

### 7.4.4 Possible next steps for this research

To improve the Ecological Model, the following should be addressed by further model development and analyses: (1) Additional endogenous stock and flow structure for grazing dynamics; (2) add structure for the effect that a low Biodiversity Index has on grazing and therefore revenue from ecotourism; (3) Policy structure needs to be triggered by decision rules that are based on monitoring the effect of biodiversity levels and restoration efforts; (4) Rates for the regime shifts should depend on a decision rule based on a variable such as "level of grazing" (e.g. when the grazing level is *x*, the rate needs to be *y*). Currently the decision rules for the occurrence of a regime shift such as changes in plant biodiversity due to changes in grazing from agricultural intensification is depicted in the model as time-dependent flows. Therefore, improvements are needed to capture structure dynamics without depending on an exogenous variable such as time. (5) Since the modelled data does not accurately reflect the oscillating patterns of the palaeo-data, a behavioural reproduction test using Theil inequality statistics could be conducted to report the mean-squared error on bias (error in mean), unequal variation (error in trend) and unequal covariation (model does not follow data exactly).

Suggestions for future research at Elandsberg PNR and the surrounding landscape include the following:

(1) Palaeo-proxy data to be calibrated with other neoecological indicators, e.g. modern pollen traps to calibrate with current plant spatial abundance, dung fungal spores with herbivore stock data, macro-charcoal with fire frequency and intensity. Thus, grazing, fire, impacts on vegetation change and trade-offs between using the landscape for endemic plant species conservation versus nature-based recreation can be quantified, and future modelling can explore these trade-offs.

(2) Incorporate stocks of other ecosystem services (water quality and soil erosion regulation) and include structure to show how changes in fire and grazing affect these stocks.

(3) As the CLD, SFD and interactive decision support tool has not yet been used as a boundary object to discuss the results with the invited participants to workshop 1, future implementation stages could involve the following: (i) Additional semi-structured interviews could be conducted for further model validation prior to additional multi-stakeholder workshops. Thus, providing an opportunity to gain insights regarding instances when there are outliers in the palaeo-data, for example, when the macro-charcoal as a proxy for local fire at Elandsberg PNR, was anomalously high in ca. AD 2004-2006 and Renosterbos decreased and grass increased. (ii) A group model workshop that presents the model results and discussing relevant scenarios. This workshop could foster collaboration through the model and focus on stakeholders' commonalities in relation to the model problem (e.g. how to increase or maintain plant biodiversity to improve ecosystem function/resilience for sustainable biodiversity conservation and ecotourism). The systems map could be used for discussion about the main causal links and identifying areas where potential interventions may be effective. Thereafter, the model user interface (Land Management Decision-Support Tool) could be used as boundary object for dialogues with conservation practitioners, commercial farmers and key land management institutions at the local governance and policy level. to demonstrate model results and conduct future scenario planning. The workshop could also demonstrate how the 'mental models' from the stakeholders have been collated and converted into the quantitative simulation model. Finally, a follow up workshop could focus on discussing strengths and weaknesses of leverage points and management interventions for sustainable land-use management.

(4) Although this fell out of the scope of the study, future research could assess the role of stakeholder engagement with the proposed conceptual meta-framework by further replication in wider CoPs, with broader coverage of ecosystem services, broader involvement of multiple stakeholder groups representing multiple sectors, and testing of different levels of intensity of participation in the model building processes, scenario planning and model validation. The proposed conceptual meta-framework, for example, could be integrated with principles from the Threshold 21 dynamic macroeconomic model, which includes environmental and social factors regarding sustainability, and has been used for development polices for many sectors (Barney et al., 1995). Furthermore, opportunities for integrated Geographic Information Systems (GIS) modelling could improve the decision-support tools (e.g. Bassi et al., 2016) for priority conservation planning in the CFR.

### 7.5 Concluding remarks for Elandsberg PNR SES

Biodiversity conservation can be enhanced with an understanding of how ecosystem services have changed over time and the feedback relationships that drives the change. Timing of a possible future regime shift is unknown, and its reversibility is uncertain, but system dynamics modelling can assist in determining this and the implications thereof. While conservation managers cannot influence climate, they could manipulate disturbances like fire and herbivory and possibly build resilience and buffer against the effects of future climate change. In conclusion, the explanation of closed feedback loops provided insight into the impacts that interactions between overgrazing and frequent fires has on system processes related to changes in plant biodiversity into the future. Simulation results from the Ecological Model show that future scenarios of land-use management policies and climate change can either hinder or benefit biodiversity conservation. The SES problem is how to increase plant biodiversity and therefore improve the ecosystem function for sustainable biodiversity conservation and climate resilience in the future. Simulation results showed that as grazing increased, Renosterbos increased, plant biodiversity decreased and fire increased. Policy analysis revealed that the system may not be able to return to its pre-1950s historical range of variability due to hysteresis and instead the land should be managed appropriately to maintain current levels of biodiversity that benefit eco-tourism and do not further degrade the landscape. Alternatively, other management options (e.g. removal of Renosterbos and restoration of vegetation units) in addition to adaptive grazing-fire management could be considered. By applying a contextual evidenced- and process-based understanding of the temporal variability and dynamic feedbacks between ecosystem services and drivers, this proof of concept showed the benefits of blending palaeoecology and PSD to explore future scenarios. The results of the combined approach can together be used to articulate safe operating parameters and management targets that can enhance social-ecological resilience and sustainability at Elandsberg PNR.

# 8 CHAPTER 8: SYNTHESIS AND CONCLUSIONS: A FRAMEWORK FOR LAND-USE MANAGEMENT DECISION-SUPPORT IN THE CFR

In this thesis, I explored a set of research objectives and components (Figure 1.2) with associated research questions and hypotheses (Table 1.1) by using mixed methods to apply the conceptual meta-framework proposed, the past-present-future lens of environmental change, in order to improve the outputs of applied palaeoecological research. Broadly, the *past* was investigated via Objective 1, the reconstruction of environmental history, Objective 2, the identification of drivers of palaeo-data and Objective 3, the interpretation of palaeo-data in terms of ecosystem services (plant biodiversity, water quality and soil erosion regulation) and resilience. Understanding the *present* included Component B as essential for considering stakeholders' perceptions of the system and past and future change, particularly the interactions between drivers, ecosystem services and resilience. Looking into the *future* was achieved via Objective 4 to model interactions and future scenarios of ecosystem processes. The research aim culminated in Component C, to improve land management decision-support", including developing insights for the identification of possible management thresholds and safe operating spaces for biodiversity conservation in the Cape Floristic Region (CFR). This was done with the overall aim of exploring the utility of palaeoecology and participatory system dynamics (PSD) modelling to provide decision-support on land-use management that maintains ecological integrity and benefits social-ecological systems (SESs).

This chapter draws together the results from applied palaeoecological and PSD in the past-present-future lens of environmental change conceptual meta-framework. For Objectives 1-3 and Component A and B, the palaeoecological results from Groot Winterhoek Wilderness Area (GWWA) and Rhenostervlei Farm sites are discussed in terms of the reconstruction of environmental change and interpretation of drivers, ecosystem services and resilience. Thereafter, for Objective 4 and Component B and C, the process-based understanding of environmental change as derived from stakeholder perceptions and modelled findings from Elandsberg Private Nature Reserve (Elandsberg PNR) are considered to increase the prescriptive nature of resilience thinking of the applied research findings (Duit et al., 2010; Herrera, 2018, 2017) (see section 2.5.1). Theoretical and practical implications for land-use policy and management decision-support as well as insights for applied palaeoecological research are highlighted. Limitations of the conceptual meta-framework are discussed as well.

## 8.1 The reconstruction of environmental change and interpretation of drivers, ecosystem services and resilience

The general context of this study is based on changes in three of four categories of ecosystem services recognised by the (Millennium Ecosystem Assessment, 2005) (Millennium Ecosystem Assessment, 2005): supporting, provisioning, and regulating. Ecological infrastructure (mountain catchment, wetlands, corridors of indigenous veg) from the three case study sites within the Berg River Catchment of the CFR co-produces ES (Cumming et al., 2014; Fischer and Eastwood, 2016). However, conserving ecological infrastructure and managing ecosystem services sustainably within the CFR is increasingly challenging without a long-term palaeo-perspective (section 2.1). The present study yields insightful findings on long-term ecological function and SES resilience and comments on whether trends in recent decades are unprecedented, still within the historical range of variability or approaching critical ecological thresholds. The sections below respond to the following research questions outlined in Table 1.1: RQ1. "*Has plant biodiversity, water quality, soil regulation, fire, herbivory and climate changed over decadal to millennial timescales?*"; RQ2. *"What anthropogenic and environmental factors have influenced/driven these changes in land cover over time, and how have the drivers interacted?"*; and RQ3. *"What do the results suggest about the resilience and sustainability of ecosystem service use at different levels of land-use intensity?"*.

### 8.1.1 Insights for the upland conservation landscape of GWWA

Exploring past climate and land-use change provides clues as to how landscapes may respond in the future. The GWWA site, as an important upland conservation site at the headwaters of the Berg River Catchment, faces management uncertainties related to the impacts on plant biodiversity and water provision from drying due to climate warming, soil erosion and the spread of invasive alien plants (IAPs), which threaten river flow. The GWWA006 record shows an initial increase in drought-resistant Fynbos shrubland abundance (*Cliffortia* pollen) with a peak during the hot and dry Mid-Holocene Altithermal (MHA) (ca. 7700-4400 BP) (section 5.5.1.2 and Figure 5.10). The hotter drier conditions were also associated with increased local and regional fire as indicated by increasing macro- and micro-charcoal.

As the climatic conditions became wetter and more humid in ca. 4495-2885 BP, plant moisture availability increased and a shift in vegetation state could suggest a change in genetic resources (Table 4.4; (Dearing et al., 2012)). As a grassier landscape developed, supporting grazing by large indigenous herbivores (LIHs) and wetter conditions resulted in increased biomass which can also enhance fire

(MacPherson et al., 2019, 2018). The increase in primary production and herbivore abundance between ca. 4495-2885 BP likely benefitted San hunter-gathers as they could take advantage of the food and fibre provided by patches of grassy-fynbos and Thicket/Forest (Table 5.1). Increased micro-charcoal is consistent with their influential presence in the region (Deacon, 1992; Meadows and Baxter, 2001; Smith, 1987) However, the impact on ecosystem services by Khoikhoi pastoralists since ca. 2000 BP was negligible at the GWWA site seen by the absence of a significant change in plant biodiversity and the local fire and grazing regime. This finding is consistent with Forbes et al., (2018)'s report of increased micro-charcoal with uncertainty regarding attribution to KhoiKhoi pastoralists and/or climate change.

Hierarchical Patch Dynamics provides an explanation for this because as climate and land-use change so does the species composition of local Fynbos and Thicket patches. The larger scale driver for ecosystem dynamics is climate and smaller scale drivers such as disturbance by fire and herbivory interact to influence the composition of the patch-mosaic landscape, affecting plant biodiversity and soil erosion regulation. An example of this is demonstrated by the oscillating trends in the abundance of Poaceae pollen and Thicket/Forest pollen types, geochemical data, coprophilous fungal spores and macro-charcoal in ca. 4495-2885 BP. As these local plant populations, that are governed by distinct landscape processes, are out of phase with one another (e.g. as seen in the quasi-stable states Zone G1, Sub-zone G2a, G2b and G2c), local patches of Fynbos and Thicket may be unstable. However, as no critical ecological threshold has been crossed the metapopulation of Winterhoek Sandstone Fynbos, which includes all vegetation elements – grasses, shrublands and thicket/forest, persists at a landscape meta-scale (Hanski, 1991; Levins, 1969; Watt, 1947; Wu and Loucks, 1995). Therefore, landscape patches provide space for recolonization by dispersal between patches which is seen in the grassy-shrubland and thicket/forest patch-mosaic landscape at present (see section 5.2).

A transitional state during ca. 2885-1610 BP was characteristic of an unprecedently lower local fire regime but the grassy-shrubland mosaic persisted with contracted thicket and forest patches confined to areas that provide climate and fire refugia (Figure 5.1). Extreme variability in sediment physical properties, Magnetics Susceptibility and geochemical indictors P, Ba, K, Sr:Ca and Zr:Rb was coupled with oscillating pattern in the aquatics and marginal:terrestrial pollen ratio (Figure 5.9 and Figure 5.10). Regional palaeoclimate records show variability associated with multiple shifts in warmer/drier and cooler/wetter climates (Table 5.1) supporting the evidence of wetting and drying conditions at the wetland. As expected, climate change was the main cause for changes in fire and herbivory that drove responses in supporting/provisioning services plant biodiversity and regulating service soil erosion regulation, at a centennial to millennial timescale (Figure 5.9). However, the ca. 10 000-year-old GWWA006 record unexpectedly shows extreme variability in local and regional fires and soil erosion regulation indicators, which explains why the GWWA landscape is heavily impacted by increased fire and erosion at present.

Although the GWWA006 record does not provide an environmental reconstruction over the last 1000 years during the periods of agricultural intensification and most lately the conservation period, the high temporal resolution and multi-proxy palaeo-perspective is still informative for understanding the drivers of change and the mechanisms of resilience including internal reorganisation and the role of vegetation-grazing-fire feedback mechanisms. The internal turnover in plant biodiversity is indicative of quasi-stable states exemplified by varying dominance of drought-resistant *Cliffortia* shrubland in the MHA transitioning to grassy-ericoid shrubland in the subsequent cooler wetter period and then to grassy-Asteraceous shrubland in the transitional late-Holocene before transitioning to a grassy drought-resistant *Cliffortia* shrubland again in the MCA (Figure 5.9). Therefore, despite extreme climatic variability, the internal reorganisation of plant functional traits shows a resilient system as the degree of return shows that the system had not reached a tipping point or critical ecological threshold and there was no change in the functioning.

It is likely that GWWA is predisposed to issues related to soil erosion regulation but anthropogenic influences such as increased burning and built infrastructure, such as roads and settlements (Table 5.1), exacerbated the effects. Although, plant biodiversity is less impacted on by land-use disturbance and more impacted on by climate change due to the remote nature of the GWWA site, the ecological character resilience of the landscape can easily be compromised given the predisposition of major fire events and the effects of wetting and drying on soil regulation in the GWWA landscape. The unprecedented increases in erosion (Zr:Rb), P and Cs associated with increased local fire at the GWWA site during the MCA is perhaps an imperfect yet reasonable analogue for the present erosion-prone landscape during increasing threats of droughts, floods and the frequency and intensity of wildfires. Specifically, the palaeo-record provides evidence for the challenges faced by management regarding soil stability associated with inappropriately placed roads since the early ca. AD 1900s and trails since ca. AD 1985 required for land-use management and ecotourism at GWWA (Table 5.1). With this in mind, I contend that without a long-term palaeoecological perspective it is risky to believe that there will always be an improvement in ecosystem integrity after perceived improved land management policies and implementation in the past. In some cases, inappropriate land-use practices such as unprecedented fire regimes, inappropriate management of extralimital game, the reintroduction of certain large indigenous herbivores (eland and bontebok), and inappropriately placed roads, may exacerbate a change in vegetation composition that was previously triggered during early European settlement or by agricultural intensification. Such unexpected findings are only possible by analysing high resolution, palaeoecological data together with historical monitored data (Table 5.1 and Table 6.1) and on a local level (Forbes et al. 2018).

### 8.1.2 Insights for the lowland agricultural landscape of Rhenostervlei Farm

Rhenostervlei Farm is a lowland agricultural site that is more heavily impacted by anthropogenic activity than the upland site but is equally important for the co-production of ecosystem services. It faces management uncertainties related to extreme weather events, wildfires and the increase in IAPs because of their impacts on ecosystem function and sustainable agricultural activities. The ca. 215-year-old RV3 record shows a decrease in the abundance of indigenous vegetation (Asteraceae *Stoebe/Elytropappus*-type, Asteraceae long-spine type-1 and *Olea* type-2 pollen) and an increase in IAPs abundance (mostly *Eucalyptus* pollen). As expected, local land-use dynamics, the implications of the Berg River management and the compounding effects of extreme weather (Table 6.1) together drove changes in vegetation dynamics. Interacting land-use and climate drivers resulted in a shift in vegetation state from indigenous trees to alien trees in ca. AD 1960s within this landscape mostly at the patch-scale seen at the riverine fringe, flood plain and the boundary between Renosterveld and Fynbos patches (see section 6.2). As expected, agricultural intensification in the 20$^{th}$ century, with associated increased fire and grazing (shocks and extreme events), decreasing plant biodiversity dramatically at the lowland site. There was a replacement of Renosterveld and Fynbos vegetation with crop cultivation, and a noticeable increase in IAPs (especially *Eucalyptus*) since ca. AD 1960s. Surprisingly, the pollen record does not show an unprecedented increase in the unpalatable shrub Renosterbos (*E. rhinocerotis*) as was evident at the lowland conservation site, Elandsberg PNR, in the ca. AD 1950s (Figure 7.7). However, land-use during agricultural intensification and the increase in IAPs have not resulted in complete ecosystem services loss as the Fynbos-Renosterveld mosaic remains a structurally resilient landscape at present.

This response in plant biodiversity was associated with vegetation clearing by burning, grazing and the spread of IAPs. Additionally, flooding and drying associated with 20$^{th}$ century climate warming influenced wetland dynamics as indicated by changes in the diatom assemblage in ca. AD 1945-1960 as the coring site is adjacent to the Berg River. Land-use intensification was the main cause for fire and herbivory which, in turn, drove the responses in ecosystem services such as decreased plant biodiversity and variability in water quality and soil regulation, at a decadal to centennial timescale (Figure 6.13). Agricultural intensification with associated increased water abstraction from the Berg River, and changes in water allocation since the construction of the weir and Voëlvlei dam in the AD 1950s-1970s, as a key part of the Western Cape Water Supply System (WCWSS), coupled with extreme weather events due to anthropogenic climate change, was expected to impact all ecosystem services negatively at the lowland site.

The high temporal resolution, multi-proxy palaeo-record indicates that although plant biodiversity is compromised, other ecosystem services such as water quality and soil erosion regulation are still intact suggesting that current land-use practices are still within a safe operating space and which allow for continued ecosystem function due to structural character resilience. Together the data suggest that a critical ecological threshold has not been crossed between ca. AD 1795-2011 though a longer-term record extending prior to European settlement would be needed to confirm this. The resilience of the system is likely due to system delays where different ecosystem services respond to shocks at different times. Therefore, current processes in this state are likely adequately regulating water purification and supporting nutrient cycling, soil formation and primary production for agricultural practices, especially during extreme climate events such as floods and the recent drought (Table 6.1). However, as plant biodiversity worsens by the invasion of *Eucalyptus*, *Pinus* and *Acacia*, ecosystem function will probably not be able to buffer threats to soil stability and water quality (see Zone 2b in Figure 6.14). With the phenomenon of the great acceleration, non-linearity and changes in system delays, a wide-spread shift to an alien-dominated state could result in the collapse of Swartland Shale Renosterveld and Atlantis Sandstone Fynbos (Figure 3.3). This is especially true for ecosystems that are already stressed or degraded, for example, by invasion by IAPs and/or intensive land-use, as they are likely to be less resilient to climate change, and therefore more susceptible to future collapse if a critical threshold is crossed (Malhi et al., 2020).

The long-term fluctuations seen in the high temporal resolution, multi-proxy palaeo-record shows evidence for increasing variance (Mottl et al., 2021) that might represent growing instability of ecological character even though it is still structurally and functionally resilient. Therefore, current and future conservation stewardship of agriculture practices should consider the ca. 215-year-old RV3 record showing the delayed negative impacts from fire and grazing as well as drying and flooding on ecosystem services that favours the expansion of IAPs and the negative consequences for natural resource management. To prevent future collapse of the system, restoration of biodiversity, monitoring and adaptive management is recommended.

## 8.2 The interface between palaeoecological and PSD: Evidence- and process-based insights for SES resilience at Elandsberg PNR, a lowland conservation site

The ca. 1300-year-old VANG palaeoecological record from Elandsberg Private Nature Reserve (PNR) showed a regime shift, crossing an ecological threshold, in plant biodiversity, local fire and herbivory during the ca. AD 1950s due to intensive grazing and burning associated with agricultural intensification. The location of Elandsberg PNR between GWWA and Rhenostervlei Farm in terms of the proposed elevation and land-use intensity gradient (section 3.2) provides a proof of concept for exploring the interface between long-term past change and potential future scenarios. The advancement of outputs from applied palaeoecological research is illustrated by using Elandsberg PNR case study (Chapter 7) to combine a long-term palaeo-perspective with participatory system dynamics (PSD) techniques in the past-present-future lens of environmental change as a conceptual meta-framework. This included the qualitative and quantitative modelling of changes in fossil pollen, macro-charcoal and coprophilous fungal spores identified in the palaeoecological record to explain feedback loops between ES and drivers and analyse future scenarios at this lowland conservation site (Objective 4).

In response to RQ-B, "How do stakeholders perceive change in landscapes and interactions between ecosystem services and drivers of change over time?" (Table 1.1), the social learning process highlighted that sustainably managing plant biodiversity as a supporting/provisioning service and herbivory as a provisioning service are just as important to agro-ecosystems as regulating services such as water quality, soil erosion and fire in order to safeguard commercial farming against future climate risk. Upon further thematic analysis of the data, there is a slight misalignment between long-term patterns and stakeholder narratives of change regarding the role of fire or grazing as the main driver of change as opposed to the key variable maintaining any given system state. The ca. 1300-year-old VANG record showed that herbivory (not fire) was the main driver of change (Forbes et al., 2018) (Figure 7.7) and the interactions between fire, grazing and Renosterbos maintained the new state as seen in the dynamic hypothesis that shows a shift in feedback loop dominance from R1 to R2 and R3 (Figure 7.9). Nevertheless, the current level of plant biodiversity found in protected areas in the lowlands, as well as fragments in agricultural landscapes, is perceived by multiple stakeholders as beneficial to the region and valued as a key part of resilience. All vegetation elements/patches are still present in both the upland conservation site, GWWA (Figure 5.1, Figure 5.9 and Figure 5.10), and the lowland agricultural site, Rhenostervlei Farm (Figure 6.1, Figure 6.13 and Figure 6.14) showing continued SES resilience at a cross-scale level, and that these are the same vegetation types observed in the respective palaeo-records.

In response to RQ4.1., *"Can a system dynamics model simulate the changes in ecosystem services and drivers identified in the palaeoecological record?"* (Table 1.1), a system dynamics model can adequately simulate palaeoecological data. As initial values in the model are set at the historical values for fossil pollen, macro-charcoal and coprophilous fungal spores from the ca. 1300-year-old VANG record, the modelled outputs sufficiently match the palaeoecological trends as historical reference modes. In response to RQ4.2., *"What are the interactions between system processes, how might these change in the future and what does this mean for sustainable land-use management?"* (Table 1.1), simulation results replicate the processes of the competitive exclusion principle shown by a shift in loop dominance from Asteraceous shrubs to unpalatable Renosterbos in Figure 7.9. The model simulated interactions between fire and grazing that reinforced the unpalatable Renosterbos-dominance since ca. AD 1950s into the future. As an ecological threshold has been crossed since the ca. AD 1950s, hysteresis makes it difficult to return to the previous alternative stable state, due to feedback mechanisms within the system. Therefore, managing grazing and fire alone at pre-1950s levels to restore to within a management threshold is not possible. However, there is no large-scale Renosterbos homogenisation and the patch-mosaic landscape comprises of varying densities of Renosterbos and post-fire ages at the landscape scale of Elandsberg PNR (Figure 3.4 and Figure 3.5) indicating that a critical ecological threshold has not yet been crossed.

The patchy and heterogenous landscape suggests that ecosystem function is not completely degraded, and ecosystem services and processes are still within a safe operating space. However, without manual clearing of Renosterbos, feedback mechanisms will favour the increase in unpalatable Renosterbos and reinforce the unprecedently high fire regime typical of the mid-20$^{th}$ century until present. An important result was comparing the average ratio of Asteraceous shrubs:Renosterbos pollen at ca. AD 750-1950 (mean of 2.1) with that of ca. AD 1950-2012 (mean of 0.4) and then modelling future scenarios until AD 2050 and AD 2100 to see whether the regime shift could be reversed (Table 7.6). The modelled results show that managing for pre-1950s levels of fire and grazing were not sufficient in decreasing Renosterbos abundance. However, maintaining the Asteraceous shrubs:Renosterbos pollen at 0.22-0.35 may allow for the current patch-mosaic landscape to persist, maintaining structural and ecosystem function within a safe operating space.

One of the rare components of this applied palaeoecological study within the CFR is the incorporation of insights derived from the social learning process (Reed et al., 2010), including the semi-formal interviews and multi-stakeholder engagement workshop with representatives from agriculture, conservation and government. SES researchers and land-use managers gained an extended temporal and process-based understanding of change (Figure 7.3) as the palaeo-evidence is interpreted in terms of ecosystem services and then discussed in terms of resilience, which are unifying themes within national and globally relevant sustainable development debates. While engaging with the qualitative models and insights from the simulation Ecological Model, stakeholders are encouraged to identify the trade-offs between ecosystem services and socio-economic benefits that will have consequences for future sustainable land-use management at the site and similar sites in the region. Future efforts are recommended to calibrate the palaeo-record with the current system state at Elandsberg PNR to be able to set management thresholds. Land-use managers can then implement adaptive ecosystem-based management to avoid the worst-case scenario (Asteraceous shrubs:Renosterbos pollen ratio of 0.08) and achieve the best-case scenario (Asteraceous shrubs:Renosterbos pollen ratio of 0.22) as shown in the future scenario analysis (Table 7.6).

## 8.3 Converging theory and practice: The past-present-future lens of environmental change considers implications for applied palaeoecology and sustainable land-use management

In using the past-present-future conceptual meta-framework advocated here, the communication of applied palaeo-research findings in a more user-friendly manner is one of the key motivations for blending a long-term palaeo-perspective, stakeholder narratives of change and SD. A status summary with a list of generative questions (Table 8.1) synthesises useful insights for advancing the applied palaeoecological field and improve management decision-support for biodiversity conservation and sustainable agriculture in the CFR. Showing the links between resilience theory, a systems perspective and the proposed conceptual meta-framework (Figure 8.1) as well as overlaying the integrative framework for understanding the dynamic behaviour of complex adaptive systems (Dearing et al., 2015, 2012) (section 2.2.2) shows the relevance of the past-present-future lens of environmental change.

Furthermore, Generative Questions (**i-vii**) in Table 8.1 adapted from Dearing et al., (2015) are intended to elicit responses from various knowledge sources, perspectives and contexts allowing for reflection on several aspects of complexity-based SESs to provide guidance and management decision-support at different levels of land-use intensity at three case study sites within the Middle Berg River Catchment. In particular, the status summary assesses the reason for why a site is important as it provides ecosystem services linked to human wellbeing values (Generative Question i; Objective 2 and Component A and B). However, land-use managers would benefit from an evidence-based understanding how the SES changed and whether ecosystem services are within the historical range of variability (Generative Question ii; Objective 1) and a process-based understanding of why the system changes and what complex dynamics and non-linear feedbacks govern threshold behaviour (Generative Question iii; Objective 4 and Component C). Grounding the evidence- and process-based understanding in theoretical principles helps advance the state of knowledge (Generative Question iv; Objective 3 and 4 and Component C); and apply these insights in the real-world by envisioning a safe operating space (Generative Question v; Component B and C), setting management thresholds and targets (Generative Question vi; Objective 4 and Component C) and monitoring indicators for adaptive management of resilient SESs (Generative Question vii; Objective 3).

Perhaps investing in research (Figure 8.1) and the compilation of an inventory of applied palaeoecological and modelling results (e.g. Table 8.1) across a wide range of spatial scales in the CFR would be well-received by resource managers (e.g. Jackson et al., (2009) and Rice et al., (2009)). Such insights relevant to sustainable land-use management will may help mainstream palaeoecology in ecosystem assessments and the national biodiversity assessment.

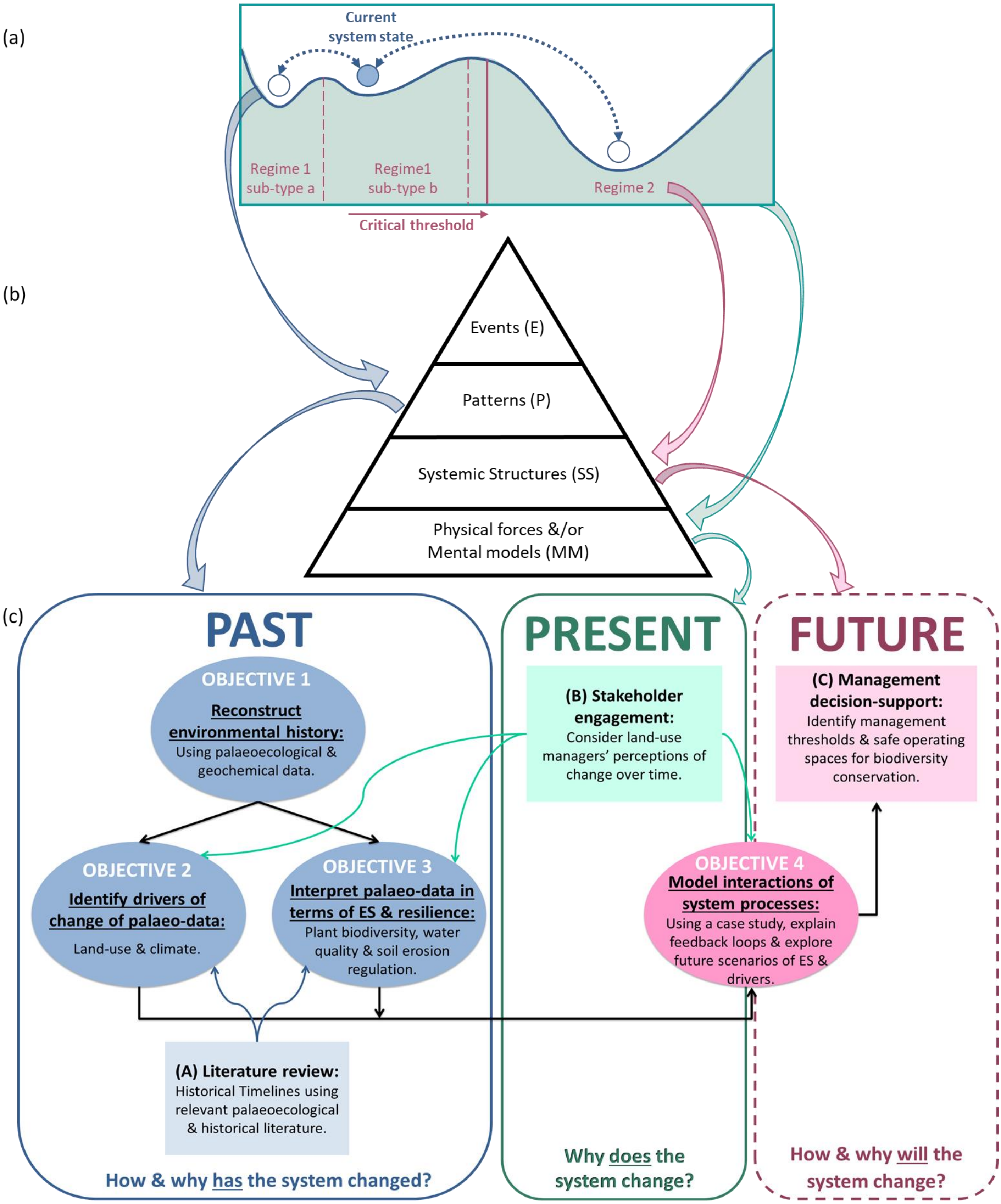



*Figure 8.1: Links between (a) resilience theory, (b) the multi-level systems perspective of the Iceberg Model and (c) the past-present-future lens of environmental change. Blue arrows represent the patterns over time, pink arrows show the regimes and thresholds are related to the systemic structures and green arrows depict the physical forces and mental models that govern the system as a whole and influence all overlaying levels (systemic structure, patters and events) of the systems perspective.*

***Table 8.1: Status summary of synthesised palaeoecological and PSD insights, with implications for land-use management decision-support at three sites within the Middle Berg River Catchment of the CFR. Generative Questions i-vii are adapted from Dearing et al. (2012, 2015) integrative framework for understanding complexity-based SESs (labelled a-g in alignment with Table 1.1). Human wellbeing values from the 2021-2030 Groot Winterhoek Complex Protected Area Management Plan (PAMP) are tentatively listed for each site.***

| **Understanding of complexity-based SESs** | **Generative Question** | **GWWA** | **Rhenostervlei Farm** | **Elandsberg PNR** | **State of the art** |
|---|---|---|---|---|---|
| (b) Interactions: Feedback mechanisms, Network functioning | **i. What ES are linked to human wellbeing values for this SES?** | Water security, environmental resilience and security from natural disasters is linked to climate and flood regulation; water and air purification; nutrient cycling; soil formation; primary production and seed dispersal. | Responsible utilisation of natural resources and security from natural disasters is linked to food and water production; climate and flood regulation; and water and air purification. | Freedom of choice and capacity to act independently, tourism and nature-based economic opportunities are linked to food and water production. | **Objective 2 and Component A and B** |
| (a) Reference conditions:, Baselines and Past analogues | **ii. Are ES within historical range of variability?** | Uncertain but surface sample from ca. 10 765-1030 BP palaeo-record is within historical range for pollen, macro- and micro-charcoal, and most geochemical indicators. | Uncertain but surface sample from ca. AD 1795-2011 palaeo-record is outside of pre-1960s range for pollen and P. | Surface sample from ca. AD 750-2012 record is outside of pre-1950s range for pollen, dung fungal spores and macro-charcoal (Forbes 2014, Forbes et al. 2018). | **Objective 1** |
| (g) Complex behaviour: Thresholds and regime shifts, Early warning signals | **iii. What are the variables that drive the regime shift maintain the state?** | Wetter conditions and more herbivory caused quasi-regime shift from drought-resistant Fynbos shrubland to grassy-Fynbos shrubland. Hypothesis: Moisture and fire maintain the regime. | Increased moisture caused regime shift from Renosterveld-Fynbos and indigenous trees to alien trees. Hypothesis: More local and regional fires and herbivory maintain the regime. | More herbivory caused regime shift from Asteraceous shrubs to unpalatable Renosterbos, and more local fires maintained the regime (Figure 7.9). | **Objective 4 and Component C** |
| (a) Baselines, (f) Fast and slow processes | **iv. What is the mechanism of SES resilience?** | *Ecological character resilience* explains functional redundancy at within-biome scale despite crossing a climatic threshold in ca. 4495 BP. | *Structural character resilience* explains functional redundancy at between-biome scale despite crossing a land-use disturbance threshold in ca. AD 1960s. | *Structural character resilience* explains functional redundancy at between-biome scale despite crossing a land-use disturbance threshold in ca. AD 1950s. Simulation results shows increased resilience of an undesirable degraded state (Figure 7.15). | **Objective 3 and 4 and Component C** |

| | | | | | |
|---|---|---|---|---|---|
| (d) Safe operating spaces | **v. Is the system currently within a safe operating space for management of SES resilience?** | Uncertain but typical Fynbos shrubland-thicket/forest patch-mosaic maintains Winterhoek Sandstone Fynbos heterogeneity. | Uncertain but Renosterveld and Fynbos remnants maintain patch-mosaic heterogeneity in agro-ecosystem. Potentially critical alien-dominated alternative stable state approaching since ca. AD 2000s. | Uncertain but grassy-shrubland-thicket patch-mosaic maintains Swartland Shale Renosterveld and Alluvium Fynbos heterogeneity. Simulation model results show potentially critical Renosterbos-dominated alternative stable state approaching since ca. AD 1950s. | **Component B and C** |
| (e) Modelling: Dynamic Behaviour, (a) Baselines, (f) Fast and slow processes | **vi. What are the proposed management recommendations?** | M&E and setting adaptive management thresholds for *biodiversity conservation and ecosystem function*. | Restoration, M&E and setting adaptive management thresholds for *conservation stewardship agriculture and ecosystem function*. | Restoration, M&E and setting adaptive management thresholds for *biodiversity conservation and ecosystem function* and managing for trade-offs between ES and socio-economic benefits. | **Objective 4 and Component C** |
| (c) Trends | **vii. What are the proposed M&E indicators for ecosystem function and SES resilience?** | Fire, soil erosion, total fynbos cover with particular attention to *Cliffortia*, restios and grass abundance. | Water quality, total Renosterveld and Fynbos cover with focus on *Olea europaea subssp. africana* and IAPs abundance. | Fire, herbivory, Asteraceous shrubs and Renosterbos abundance (Forbes 2014, Forbes et al. 2018). | **Objective 3** |

### 8.3.1 Resilience thinking in complexity-based SESs

#### *8.3.1.1 Interconnectedness and complexity thinking*

Functional traits, climatic and geographical envelopes are critical factors in the interconnectedness between plant community assembly and ecosystem function at multiple ecological temporal and spatial scales (Chapin et al., 2008; Slingsby et al., 2014). As there is interconnectedness between plant community assembly and ecosystem function (Chapin et al., 2008, 1997; Slingsby et al., 2014), there is a ‘grey area’ at ecotones where different species with similar traits can both occur at the between-biome scale. An example of this is the statistical significance in the RV3 pollen zonation not equating to a regime shift. As *Eucalyptus* and *Acacia* and thicket/forest taxa have similar biotic functional traits they may have similar ecosystem processes and function. Therefore, although alien trees increased since ca. AD 1960s structural character resilience ensured that an ecological threshold has not been crossed at the site. However, the dramatic increase in *Eucalyptus* pollen since early ca. AD 2000s is cause for concern that ecological degradation is happening at a rapid rate.

Increased rates of change are a general global phenomenon in plant biodiversity and Mottl et al., (2021) report that the rates of change accelerated strikingly during the late Holocene (ca. 4 600-2 900 BP). The long-term palaeo-perspective in the present study confirms increased rates of change that might insinuate growing instability of ecological character over time and into the future. The decrease in time between changes in GWWA pollen zonation seen by the initial change occurring after ca. 6270 years, then ca. 1610 years and the last change after ca. 1275 years (Table 5.4), may be an indication of acceleration in the rates of change. This finding questions the maintenance of SES resilience in the area. Furthermore, the change in the Rhenostervlei Farm system delays (Table 6.4) may be an indication that current SES function and resilience is not sustainable in the medium-long-term future. This finding is consistent with research that shows long-term and delayed impacts of IAPs on Fynbos plant diversity, with existing evidence for the genitive effects of altered biodiversity composition on ecosystem functions and ecosystem services (Slingsby et al., 2017, 2014).

#### *8.3.1.2 Mechanisms of resilience within- and between-biome scales*

Detailed descriptions of each site are provided in the preceding section, here I focus on the mechanisms that confer resilience at different scales and embed this in resilience and complexity thinking. At GWWA, ecological character resilience to climate change was conferred through internal reorganisation within Fynbos vegetation. At Rhenostervlei Farm, we see functional redundancy and ecosystem services provision despite the increase in alien species and relatively intensive land use, consistent with structural character resilience. At Elandsberg PNR, there was a transition to a more Renosterbos dominated state driven by increased grazing and maintained by unprecedented levels of fire and grazing, but all landscape elements/patches were retained. These different manifestations of resilience can be linked through theoretical concepts such as functional redundancy, quasi-stable states, ecological character resilience and structural character.

As exemplified by the upland conservation site, GWWA, although an ecological threshold has been crossed in the past in ca. 4495 BP, the multi-proxy data shows that the transition was not critical to Fynbos resilience as climatic change was buffered through internal reorganisation. Therefore, the quasi-stable states and maintenance of the grassy-shrubland and thicket/forest mosaic at present indicates that past and current levels of plant biodiversity were historically and are currently managed within a safe operating space. As the GWWA site is not located at an ecotone, ecological character resilience seems feasible in the long-term as the likelihood of a transition to Thicket/Forest is minimal and functional redundancy in fynbos confers resilience at the biome level. Thus, crossing a critical threshold and a regime shift to a predominantly grassy landscape is unlikely (Curtis, 2013; Forbes, 2014; Newton, 2008) as functional redundancy in Fynbos confers resilience at the biome scale. However, further simulation experiments are recommended to test this and qualify this hypothesis. Furthermore, the high temporal resolution, multiple palaeo-proxy analysis of plant biodiversity, water quality and soil erosion regulation at Rhenostervlei Farm provides evidence of increased land-use disturbance but ecosystem function is maintained through structural character resilience. However, to maintain ecosystem function water quality monitoring should be included in future land-use planning and practice at this site and the surrounds. In the case of Elandsberg PNR, a regime shift occurred as suggested by increased Renosterbos abundance due to increasing herbivory and fire since ca. AD 1950s. Until present a critical ecological threshold has not been crossed as all elements of the patch-mosaic landscape are still present, which allows for continued SES resilience. The patch-mosaic landscape contains varying post-fire ages and ecotourism benefits associated with game farming.

In complexity terms, the slower processes are viewed as controlling resilience (Biggs et al., 2009; Carpenter et al., 2009). Changes in trends or patterns (Figure 8.1) of slower processes are not clearly discerned in the relatively short timescales offered by neoecological monitoring records. This is substantiated by findings from the Elandsberg PNR as a proof of concept (Figure 7.7). The ecological processes changed slowly pre-1950s, particularly the slow increases in herbivory since the ca AD 1800s coupled with low levels of fire, these controlled the resilience of a patch-mosaic landscape dominated by Asteraceous shrubs. After the ca. A 1950s regime shift the rates of change increased for herbivory, and a slower increase in local fire controlled the more resilient patch-mosaic landscape that has increased Renosterbos-dominance. Therefore, and important insight from the present study that used mixed methods to quantify changes in ecosystem services over time is that regime shifts at ecotones, are difficult to reverse by only manipulating land-use drivers that cause and maintain alternative stable states. As the rates of change in drivers increased the landscape processes increased and resilience in a particular vegetation state is no longer maintained. The new vegetation state is therefore governed by new slow processes, for example, Figure 7.9 shows the slow process of Renosterbos homogenisation by reinforcing loop R3 (Unpalatable shrub causes fire persistence) which is currently maintaining the resilience of an undesirable state.

Another important mechanism of resilience is internal reorganisation, which allows Fynbos to persist in alternate quasi-stable states, as seen at GWWA and in previous work by MacPherson et al., (2019). Underpinning this adaptability in fynbos is the vast diversity which confers resilience through functional redundancy (Gillson et al., 2020; Hunter, 2011; Jia and Whalen, 2020; Oliver et al., 2015; Yachi and Loreau, 1999). I propose that functional redundancy in can be explained in two ways when maintaining SES resilience in the present study: (1) ecological character resilience or (2) structural character resilience. Firstly, functional redundancy by ecological character resilience can be recognized by a quasi-regime shift in a metastable vegetation type with an ecological threshold being crossed. An example is the quasi-stable state transition of the grassy-shrubland-thicket/forest mosaic landscape at GWWA after crossing a climatic threshold in ca. 4495 BP. The quasi-regime shift from drought-resilient Fynbos shrubland to grassy-Fynbos shrubland was characterised by distinctive grass co-dominance with varying dominance of drought-sensitive ericoid, Asteraceous, and drought-resistant *Cliffortia* shrubland during ca. 4495-1030 BP.

Secondly, functional redundancy by structural character resilience can be recognized by a loss of original ecological character due to instability of a vegetation type, with an ecological threshold being crossed. An example of this is the regime shifts that occurred in the mid-20$^{th}$ century at the lowland sites. At Rhenostervlei Farm alien trees started to replace the function of indigenous trees since the ca. AD 1960s. At Elandsberg PNR unpalatable Renosterbos replaced the function of other Asteraceous shrubs since ca. AD 1950s. Structural character resilience allows other regulating and provisioning services such as water quality, soil regulation and ecotourism related to game farming to remain intact despite the partial ecological degradation through loss of “natural ecological character” (Davidson, 2016; Finlayson et al., 2005; Gell et al., 2013, 2018). Furthermore, Rhenostervlei Farm still has irreplaceable fragments of Renosterveld and Fynbos (section 3.2.2) and Elandsberg PNR has the largest area of intact West-Coast Renosterveld in the region (section 3.2.3). These fragments offer a key supporting and provisioning ecosystem service, nestled between croplands and grazing lands that provide food and fiber. Land-use management planning would benefit from palaeo-evidence to set management thresholds such as TPCs/LACs to prevent a future regime shifts to more resilient degraded alternative stable states dominated by unpalatable shrubs and IAPs. In turn this would support the protection of other abiotic ecosystem services such as water quality and soil erosion regulation (Figure 7.8).

It is possible that structural character resilience and system delays to shocks and extreme events avoid complete ecological degradation (Generative Question iv in Table 8.1). Different ecosystem services responding to the same driver at different times is possibly ensuring functional redundancy in West-coast Renosterveld and Fynbos fragments of the lowlands. However, regime shifts from the mid-20$^{th}$ century or possibly in the near future may affect ecosystem function which is currently protecting the unique plant biodiversity of the CFR. Changes in ecological character at both lowland sites, Rhenostervlei Farm and Elandsberg PNR, will have consequences for land-use management including knock on socio-economics effects if future scenario planning and systemic implementation ignores the implications. Trade-offs between ecosystem services that benefit biodiversity conservation as opposed to other sectors (Generative Question vi in Table 8.1) must be considered together with practical steps to preserve the integrity of multiple ecosystem services such as water quality, soil regulation, herbivory and fire, to minimise further implications for plant biodiversity at a catchment scale to maintain health of the multi-functional CFR.

#### *8.3.1.3 Ecological and management thresholds, and scenario planning*

Using our long-term palaeoecological perspective allows for an evidence-based understanding of state changes over time, in the past and imagining the future. The strength of this study is the reconstruction of environmental history using high temporal resolution, multiple palaeo-proxies drivers that extends the timescales of our knowledge on past ranges of variability of plant biodiversity, water quality, soil erosion regulation, fire, grazing and climate. Further participatory efforts are needed to develop management thresholds by calibrating the pollen record with quantitative estimates of land cover of the proposed indicator taxa (e.g. Generative Question vii in Table 8.1). In this regard, collaboration with stakeholders about what management thresholds are practicable in terms of different stakeholders preferred ecological state, within what is ecologically feasible (Biggs et al., 2011; Biggs and Rogers, 2003; Gillson and Marchant, 2014; McLoughlin et al., 2011; Rogers, 2003). Furthermore, to manage for climate and SES resilience, we need to start planning for known unknowns such as droughts, floods and fires, which will mean applying our knowledge of past ecosystem resilience to possible future scenarios.

The use of long-term series data for the identification of past regime shifts by monitoring and evaluation (M&E) of indicators and whether the current state is within a safe operating space for SES resilience (e.g. Table 8.1). Therefore, with calibration and stakeholder collaboration in place (e.g. Gillson and Duffin, 2007), insights from the present study could contribute to developing management thresholds (e.g. see historical range of variability in Figure 5.11 and Figure 6.15), though they are not in themselves management thresholds.

As a lowland SES, Elandsberg PNR as a for structured and operational (Clifford-Holmes et al., 2017b; Richmond, 1993) (section 2.4.1), therefore adaptive management is essential (e.g. management thresholds in section 2.5). Potential ecological thresholds and management recommendations such as pre-1950s fire and grazing levels (Forbes et al., 2018) were tested using the SD simulation model. Model results show that decreasing Renosterbos is a slow process (Biggs et al., 2009; Carpenter et al., 2009) if not removed by manual clearing and therefore may not be a feasible management option. Therefore, adaptive grazing-fire management and M&E is required to maintain current levels of plant biodiversity and ecosystem function at the site in future to avoid an undesirable critical ecological threshold being crossed in future (section 7.4 and Table 8.1). Notwithstanding the possibly more resilient undesirable state (Grace and Pope, 2015) dominated by unpalatable Renosterbos, successfully managing for SES resilience is still possible when protecting nature and people, and climate change adaptation are considered via ecosystem-based management and adaptation (DEA and SANBI, 2016; DEFF, 2019; Gann et al., 2019; O'Higgins et al., 2020; Pasquini and Cowling, 2015) (section 2.5.1). Thus, SES resilience cannot be done without the acknowledgement of ecosystem services trade-offs and socio-economic benefits outside of plant biodiversity conservation priorities and therefore need to be factored into participatory scenario planning and setting TPCs in future.

In the case of the GWWA site, a millennial timescale provided a long-term overview to determine that a regime shift had taken place due to climate and fire variability in the absence of significant anthropogenic influence. The GWWA006 palaeo-record provides evidence for 'natural ecological character', 'limits of acceptable change', and clues of 'degraded state' of this wetland, and can therefore inform dialogue on how to manage changes in ecological character at GWWA since it has implications for managing water flows downstream throughout the Berg River Catchment. Managing GWWA appropriately into the future is essential since the Groot Winterhoek Mountains are a national strategic water source area in South Africa and therefore an important ‘water tower’ for the Western Cape.

Without a precolonial reference condition for the Rhenostervlei Farm site (Forbes et al., 2018; Gillson, 2015; Manzano et al., 2019; Pauly, 1995; Wolfe et al., 2012), it is uncertain whether a regime shift has already taken place at Rhenostervlei Farm prior to ca. AD 1795 or after the sediment core was extracted in AD 2011. However, the modern vegetation survey in 2020 suggests that the ecosystem is still functional because all vegetation patches are in the landscape as well as the RV3 record. . In the case of the Elandsberg PNR site, modelled results show that ecotonal landscapes may be more susceptible to moving to an undesirable state that is even more resilient. The 1300-year-old VANG record provided a pre-colonial benchmark that helped to determine whether a regime shift in fire, grazing and Renosterbos abundance had taken place. The long-term palaeo-perspective was enhanced by a process-based understanding of the interconnectedness between ecosystem services and drivers of change. An example of this is the future scenario of Renosterbos homogenisation at Elandsberg PNR. Without the simulation modelled results, the reversibility of the regime shift was unknown (Forbes et al., 2018) but this study suggests hysteretic behaviour, therefore, shows the difficulty in returning to a pre-1950s state (Figure 7.14 and Figure 7.8).

This descriptive usage of thresholds for resilience concept (Table 8.1) is all well and good, but the insights still only offer a speculative idea of the future at GWWA, Rhenostervlei Farm and the surrounds. To manage for climate and SES resilience, we need to start planning for future unknowns such as droughts, floods and fires. In the future, formally protected areas such as Elandsberg PNR and GWWA and agricultural lands such as Rhenostervlei Farm can undergo further regime shifts by crossing critical ecological thresholds. The management thresholds developed by calibrating the pollen record and collaboration with stakeholders (see above) could explore future scenarios where extreme events or continued degradation drive ecosystems across critical thresholds. The blending of palaeoecology and PSD is heeding an urgent recommendation to move from a descriptive to a prescriptive understanding of resilience (Dearing et al., 2019, 2014, 2012, 2011; Herrera, 2018, 2017; Jeffers et al., 2015; Mckay et al., 2016; Scholes, 2017) that can be applied in exploring future scenarios.

The development of simulation experiments such as the construction of stock and flow diagram (SFD) to explain the essential dynamics of ecosystem services and drivers that could provide additional insights on ecological degradation at these sites. New causal loop diagrams (CLDs) and SFDs that capture feedbacks for GWWA and Rhenostervlei Farm are beyond the scope of the present study. However, in their absence, I make the case for addressing diagnostic and generative questions within the status summary about the ecosystem character (states), processes, services and function which is both relevant to the management of SESs and easier to address when set within the conceptual meta-framework (Table 8.1). The key aim of ecological restoration is to the manage for the capacity to recover to the intact undegraded state by overcoming resistance or negative resilience and strengthening positive SES resilience (Generative Question vi, Table 8.1). Some suggested management strategies and socio-economic and environmental scenarios that could be analysed when planning for restoration and conservation include:

- **Climate-resilient erosion prevention scenarios at GWWA:** In using SD terminology, if the current 'base case' conditions continue, will the current human and financial investment in erosion prevention be effective when unpleasant surprises such as droughts, floods and uncontrolled wildfires occur? In this case, Sub-zone G2c patterns and trends which are graphs over time can be used as the 'reference modes', and the reference conditions for ecosystem services and drivers during the MHA and MCA as past analogues (Figure 5.10) for policy and scenario analysis in a simulation experiment.
- **Climate-resilient river rehabilitation scenarios at Rhenostervlei Farm:** If alien trees such as *Eucalyptus*, *Pinus* and *Acacia* are cleared and replaced with local thicket/forest trees, to what extent will extreme weather such as droughts and flooding, uncontrolled wildfires and extreme land-use such as overgrazing impact on water quality and current agricultural practices? In this case, patterns and trends of Pollen Sub-zone R1a and Diatom Sub-zone A (Figure 6.14), can be used as the 'reference modes' in a simulation experiment and various system delays can be tested under multiple policy and future scenarios.

Perhaps system delays and functional redundancy can contribute to continued resilience in patch-mosaic landscapes, buy us some years before complete ecological degradation has negative consequences. However, urgent collaborative efforts with multiple stakeholders to prevent unsustainable land-use management practices is needed. The degree to which fire or grazing drives or maintains a state may also be spatially and temporally dependent so in addition to experimental research and adaptive management, further integrated palaeoecological and PSD research is needed to determine the effects and interactions between different fire seasonality and grazing regimes.

Although the combination of paleoecology and PSD is novel in South Africa and still under development globally, this study proves that blending palaeo-data, stakeholder perceptions and system dynamics is possible. Land-use managers were enlightened by the prospect of the proof of concept and that the approach can provide site specific decision-support. However, by adding more studies to upscale to the catchment level and adding more sites to downscale for site-specific recommendations would be best practice for theory and implementation (Scholes, 2017). With increased studies applying the past-present-future lens on environmental change and on sustainability, it may be feasible to also develop generic decision-support tools for integrated catchment management at a greater spatial scale.

### 8.3.2 End-products and learning insights

Rooted within the ethos and fundamental principles of applied palaeoecology (Birks, 1996) this study presents a management decision-support tool in the format of a free user interface developed in Stella Architect. The intention is for various stakeholders to be able to engage with the dynamic content in an explicit way and for further engagement to facilitate a thinking environment so that land-use managers can develop insights that will inform their management and governance of natural resources. Moreover, a decision-support toolkit includes a selection of outputs from this study: the Storylines and Scenarios that uses insights from both palaeo and PSD (Figure 7.8), Status summary on SES resilience (Table 8.1), Stella Architect Land Management Decision-Support Tool (Figure 7.16) (https://exchange.iseesystems.com/public/cherie-dirk/ecological-model-land-management-decision-support-tool), and the methodological research process steps (Table 4.5 and Figure 8.2). Together these end-products may be valuable during future multi-stakeholder engagement and participatory scenario planning workshops within and outside of palaeoecology and biodiversity conservation networks (see section 2.5.3 in Literature Review Chapter 2). Adequate follow-through of applied palaeoecological studies means that the findings are also policy relevant and that there is reflection processes built into the research design (e.g. multi-stakeholder engagement workshop and iterative five phases of the SD methodology), with a transparent documentation of research process steps (Table 4.5).

Navigating dimensions of the past-present-future lens requires several integrative process steps to gain a better understanding of change over time and SES resilience (Figure 8.1). Sharing research process steps promotes learning but they need to be operational and functionally useful (Biggs et al., 2008) for boundary crossing. Figure 8.2 further highlights the motivations for modelling (Clifford-Holmes et al., 2017b) (see section 2.4.1). The integrated analyses were conducted to better understand long-term change (continuum 2), the interdisciplinary research design endeavoured to provide recommendations for policy-makers and land-use managers (continuum 1). The research approach also included mediated modelling and knowledge brokering in the continued effort towards a social learning process during future multi-stakeholder engagement and therefore move towards intervention (continuum 4).

The translation of palaeo-data into useful metrics and end-products encourages knowledge brokering to inform restoration and adaptive ecosystem-based management in conservation and agricultural landscapes along the elevation and land-use intensity gradient in the Middle Berg River Catchment, CFR and South Africa. Therefore, the fluidity of boundary crossing (grey area in Figure 8.2) has the potential to propel knowledge brokering to better determine synergies and trade-offs between natural resources, and this is where future innovative inter- and transdisciplinary SES research should focus. As the stakeholder engagement component of this study was originally designed as a social learning process, follow-up with land-use managers is required. An assessment of feedback from stakeholders on the process and the end-products would show whether a change in understanding has taken place in the individuals involved and becomes situated within wider social units or communities of practice (e.g. Step 4 of Figure 2.4) (Brent et al., 2017b; Clifford-Holmes et al., 2017b; Reed et al., 2010; Walker et al., 2002).

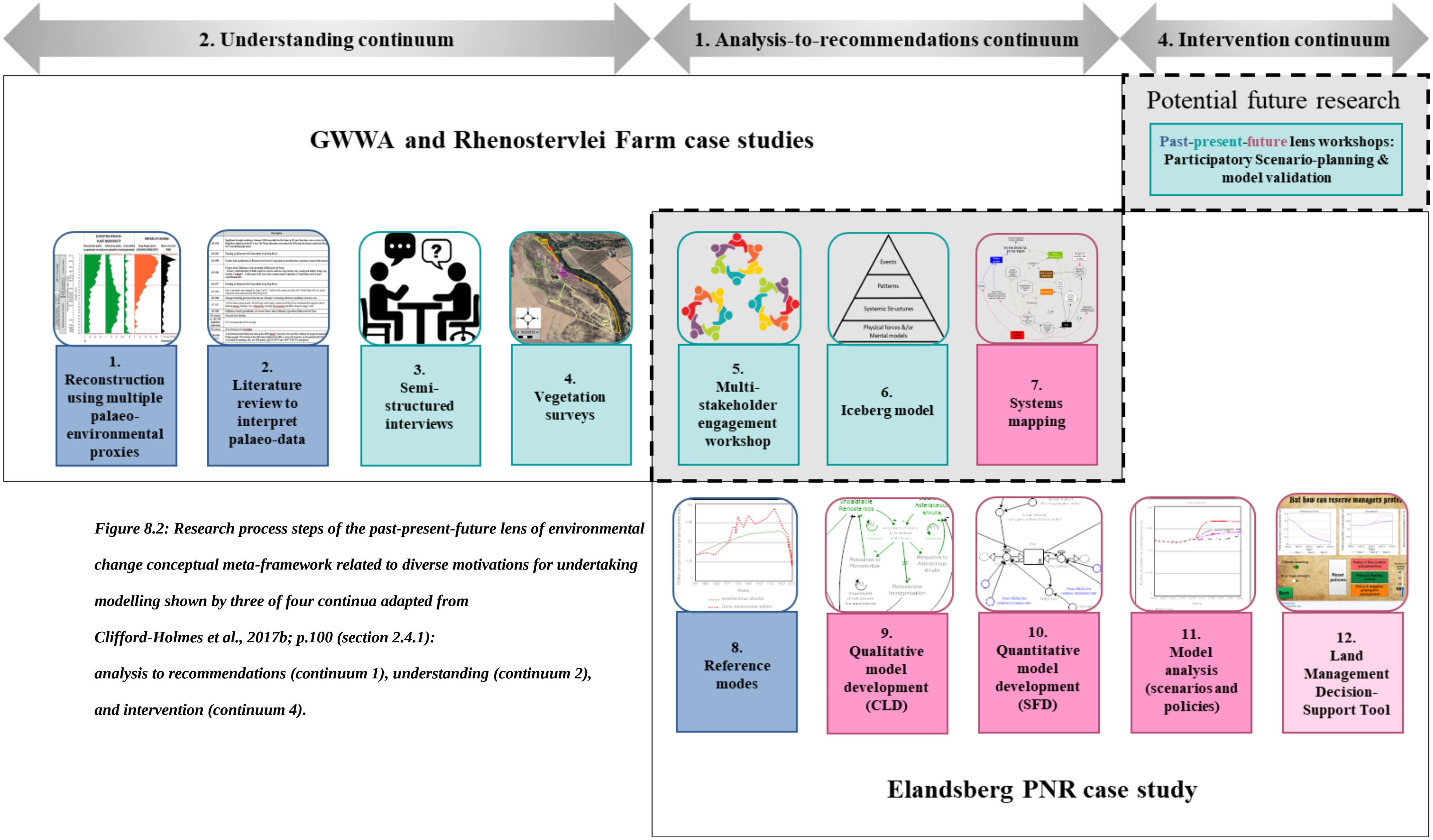



*Figure 8.2: Research process steps of the past-present-future lens of environmental change conceptual meta-framework related to diverse motivations for undertaking modelling shown by three of four continua adapted from Clifford-Holmes et al., 2017b; p.100 (section 2.4.1): analysis to recommendations (continuum 1), understanding (continuum 2), and intervention (continuum 4).*

### 8.4 Limitations of the study and ways forward to improve an evidence-based and process-based understanding of SES resilience

In addition to revealing log-term ecological surprises, applied palaeo-investigations offer an opportunity to identify some research limitations that can be addressed to improve applied research in the future. Given the temporal scales of the GWWA006 and RV3 records a limitation of this study is that the RV3 record could not reconstruct environmental history during the MCA and both the GWWA and RV3 could not reconstruct environmental history during the cold and wet Little Ice Age (LIA) dated ca. AD 1400-1800. Furthermore, this study cannot comment on whether land-use by European settlers would have the expected low to medium impact on biodiversity, water quality and soil erosion regulation and fire and herbivory regimes. Additional palaeoecological investigation to explore these impacts within the uplands of Middle Berg River Catchment is required to determine ecologically realistic reference conditions for maintaining SES resilience particularly to analyse long-term data that suggests a resilient state that was maintained by low levels of grazing and fire during drier climatic conditions (see section 6.6.1.1).

Furthermore, though Fynbos is incredibly diverse, much of the pollen can only be resolved to family or at best genus level, which makes pollen counting manageable (see section 5.5.1.1, Figure 5.3 and Figure 5.4) but limits ecological interpretation. In this case, the taxonomic resolution of pollen data does not permit examination of turnover at the species or genus level. Therefore, the present claims on functional redundancy at the case study sites are incomplete and further mixed methods analyses that adopt a pluralistic, social-ecological view would be beneficial (Holden, 2017; Sellberg et al., 2021). The rarity of applied palaeoecological datasets in the Berg River Catchment is arguably a limiting factor in mainstreaming palaeoecology in South Africa (Figure 2.2) and this study shows the benefits of a long-term palaeo-perspective.

Another important condition that limits the application of the proposed conceptual meta-framework is that it does not benefit all sites within the CFR. As many plant species are rare with 68% endemism, the protection of such unique biodiversity at any given site may be a higher priority than the identification of past regime shifts and management thresholds for the maintenance or increase of ecosystem function. It was not within the scope of the present study to calibrate the GWWA and RV3 records with current vegetation cover nor to collaborate with the relevant site-specific stakeholders to decide on management thresholds to be implemented. Furthermore, quantification of palaeo-proxies into ecosystem services indices (e.g. Dearing et al. 2012) is another crucial step needed towards using references conditions to inform management targets. However, further research in both these regards is highly encouraged, especially since there are trade-offs with plant biodiversity conservation and other ecosystem services (Dearing et al., 2019), as well as trade-offs between ecosystem services and socio-economic benefits such as game management and agriculture (Armstrong McKay et al., 2019) that generate revenue and contribute directly to the South African economy (section 2.4.1 and 2.5.1). For example, at GWWA, there has been an increasing issue related to soil erosion due to improperly placed roads and trails required for both management access and tourism (Wheeler 2016 pers. Comm.). At Elandsberg PNR, high levels of herbivory and fires during agricultural intensification were exacerbated after the reintroduction of LIHs since ca. AD 1970s (Appendix Table 10.8). Game management and other forms of ecotourism that are botanically focused contribute to the tourist attractions that generate significant revenue in private nature reserves (Figure 7.4) (Clements et al., 2020; Forbes et al., 2018; Holden et al., 2021; Shumba et al., 2020, 2021). The lack of human and financial resources to support for the biodiversity sector affects the management of alien invasive plants, fires and roads, thus undermining ecological infrastructure in the long-term.

Although ecosystem services and SES resilience may be regarded as indirect variables in non-biodiversity sectors such as agriculture and urban planning, recognition of their importance for both food security (Bennich et al., 2018; Herrera de Leon and Kopainsky, 2019) (section 2.4.1) and biodiversity conservation (Armstrong McKay et al., 2019; Dearing et al., 2015, 2012; Holden et al., 2021; Mottl et al., 2021) (section 2.2.2) is increasing, especially in the CFR lowlands where conservation of unique plant biodiversity is more urgent and critical (Forbes et al., 2018; Roberts et al., 2001). However, in this study only a selected number of ecosystem services and drivers were analysed with a wealth of other palaeo-proxies for ecosystem services to investigate further (see section 2.2.1). Additional analyses on other ecosystems services should include soil fertility, bio-control, ecotourism and agricultural yield with associated profits crucial to people’s livelihoods (Figure 10.24). This would ensure that the insights are applicable for policy- and decision-makers who must consider both ecological and the socio-economic factors of resilience and sustainability. With regard to ecosystem services and management, given the multi-functionality of the CFR landscape, there are different stakeholder groups from multiple sectors. Multiple stakeholders' priorities might differ (Table 4.5; Step 3, 5 and 6) and as a result the description of the desired situation for sustainable land-use management will be nuanced based on how different stakeholders perceive the SES problems. Decisions for restoration and natural resource management have social, economic and multi-level governance dimensions (Cilliers et al., 2013; Helfgott, 2018; Roux et al., 2017), particularly when implementation requires decentralized and focused efforts (both top-down and bottom-up). Therefore, ways that encourage stakeholder participation and buy-in, and that can test ecological function and resilience by future scenario and policy analyses is key.

### 8.5 Conclusion

This thesis applies a conceptual meta-framework, *the past-present-future lens of environmental chang*e, that is imbedded in systems-thinking, complexity-based SES and resilience theory. This thesis serves as a chronicle of a methodological approach that blends insights from palaeoecology and participatory system dynamics (PSD) using a framework of resilience theory. It documents and shares methodological process steps, ensuring transparency and an opportunity to improve applied palaeoecological research in South Africa. Even more relevant is the decision-support toolkit comprised of end-products that blending palaeoecology and modelling to potentially contribute to sustainable management.

A healthy multi-functional CFR ecosystem that is climate resilient is the desired situation that will ensure SES resilience and sustainable land-use. This study explored conservation areas as well as productive, economically viable agricultural lands essential for food security that co-produce ecosystem services at a landscape/catchment level. To maintain or restore ecosystem integrity, promote good natural resource governance and climate change resilience, policy and land-use management decision-support can be improved by the application of the conceptual meta-framework. The benefits of using palaeoecology and PSD include the identification of reference conditions and states, the exploration of historical ranges of variability, the mechanisms that confer resilience (Chapter 5 and 6), and the simulation experiment to test recommendations for management thresholds and targets (Chapter 7).

At the upland conservation site, GWWA, which is at a higher elevation and experiences lower levels of land-use disturbance, past warm periods and shifts in the dominance of different elements of the vegetation provided insights into resilience and response to climate change, and further highlighted the vulnerability of this ecosystem to the risk of soil erosion and the relative abundance of grass and *Cliffortia*. At the lowland agricultural site, Rhenostervlei Farm, which is at the lower elevation and has a history of more intensive land-use disturbance, past agricultural intensification interacting with climate variability resulted in shifts in the dominance of indigenous vegetation to invasive alien vegetation and provided insights into structural resilience and functional redundancy. This further highlighted the vulnerability of this ecosystem to the degradation of ecosystem services, particularly decreased water quality, *Olea europaea subssp. africana* and other Thicket species and increased IAPs abundance such as *Eucalyptus* and *Acacia* spp. At the lowland conservation site, Elandsberg PNR, which is at mid-elevation and has intermediate levels of land-use disturbance, past agricultural intensification including an increase in grazing resulted in a state shift in the dominance of other palatable Asteraceous shrubs to unpalatable Renosterbos. PSD results highlighted that hysteresis would make the reversibility of the regime shift difficult due to non-linear feedback mechanisms within the system. This further highlighted that the current system state is undesirably resilient (or “resistant”) and crossing a critical threshold to a Renosterbos-dominated alternative stable state in future would increase vulnerability of the SES to the degradation of ecosystem services, particularly decreased plant biodiversity, increased Renosterbos abundance and landscape homogenisation.

A generic recommendation would be that land-use managers consider climate-resilient reference condition/states from the past, try to keep all landscape elements and heterogeneity as this helps to confer resilience, maintain plant biodiversity and other ecosystem services. The MHA and MCA could be useful past analogues in helping land-use managers to decide whether current levels of fire and abundance of key plant taxa are within acceptable limits. Applying the past-present-future lens of environmental change can contribute to address sustainability, resilience and SES problems. Through this lens, innovation and development is driven with a focus on identifying and activating leverage points to mainstream palaeoecological insights into multiple institutionalised governance levels of the local, regional and national scale.

The applied palaeoecological approach should aim to actively build reflexivity into the research process and ensure opportunities for knowledge brokering so that other researchers and practitioners can tailor their work for appropriate, context-based interventions. Reflexive outputs such as lessons learnt and conceptual frameworks could be compiled in the format of policy briefs, decision-support toolkits and grey literature and shared with other relevant communities of practice (CoPs). The generation of palaeoecological outputs needs to consider the end-user and the applicability of the information to effectively inform sustainable land management through engagement with policy relevant concepts such as ecosystem services, ecosystem-based management, EbA, ecological restoration and the SDGs. In conclusion, it is emphasised that the maintenance of ecosystem function of conservation and agricultural landscapes that co-produce ecosystem services in the Middle Berg River Catchment of the CFR are a task for land-use managers in collaboration with scientists from multiple disciplines that use systems thinking as a framing backdrop to implement sustainable land-use management important in integrated catchment management.

The hope is that through my research, a long-term, palaeo-perspective will be on the radar of key decision-makers (public and private land-use managers, and policy-makers) when it comes to our country’s natural resources – particularly water quality, erosion regulation, fire and herbivory management, and biodiversity conservation in the Berg River Catchment. By being mindful of the long-term perspective of ecosystem services, local stakeholders will be better equipped (responsive, effective and efficient) to identify management thresholds under future climate scenarios in order to adapt to climate change and safeguard social, ecological and economic resilience.

## 9 REFERENCES

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# 10 APPENDICES

## 10.1 Principles for coarsely calibrating the modern pollen with the resent vegetation

The uniformitarian approach, which suggests that the fossils have the same biological and environmental requirements as the modern-day counterparts, is often applied when reconstructing environmental history from stratigraphical records. The aim and objectives of the present study do not include formal calibration between archival and modern states such as pollen and charcoal (Julier et al., 2021; Maezumi et al., 2021), therefore, the vegetation survey data was coarsely calibrated with the modern pollen rain as follows:

1. The number of pollen taxa that were identified during the modern vegetation surveys (BB relevés) was compared with the surface sample pollen to determine whether there was significant overlap in the pollen taxa, usually to family and/or sometimes genus level. If there is an overlap in taxa identified, then the surface sample pollen is considered a good representation of the current vegetation assemblage.
2. If the modern pollen rain successfully represents the current vegetation, then the fossil pollen accumulated in the sediment can be used to reconstruct environmental history, particularly how plant biodiversity has changed in response to changes in climate and land-use.

Thus, by transferring an organism's modern ecological situation to its fossilised occurrences within the palaeo-record, one can create a snapshot of the environment at the time of deposition and with it resolve external environmental factors relating to climate and distribution.

## 10.2 Datasheet for vegetation surveys

### BRAUN BLANQUET (BB) RELEVÉS

| **Site:** | | **Date:** | **Plot** | **1** | **2** | **3** | **4** |
|---|---|---|---|---|---|---|---|
| **Vegetation unit and description** | ***e.g. Alluvium Fynbos:*** *Ring of Alluvium Fynbos on the ridge north and east of the wetland.* | | **Coordinates** | | | | |
| | | | **m.asl** | | | | |
| | | | **Slope** | | | | |
| | | | **Aspect** | | | | |
| | | | **Area ($m^2$)** | | | | |
| | | | **Last Fire** | | | | |
| **Collector's No.** | **Photo ID No.** | **Species identification** | | | | | |
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| | | | | | | | |
| | | **Dung counts** | | | | | |
| | | **Bare ground** | | | | | |

| **ADDITIONAL INFORMATION ABOUT THE PLOTS (BB relevés/analysis)** |
|---|
| **Plot 1** |
| **Plot 2** |
| **Plot 3** |
| **Plot 4** |

## 10.3 Detailed information on laboratory methods

### 10.3.1 Details of chronology and age depth modelling of sediment cores

#### *10.3.1.1 Accelerator mass spectrometry (AMS) radiocarbon dating*

Accelerator mass spectrometry (AMS) radiocarbon dating is one of the earliest and most widely used of the radiometric techniques. Carbon-14 ($^{14}C$) is a radioactive isotope of carbon which eventually decays (via the emission of beta (β) particles) to form the stable element Nitrogen-14 ($^{14}N$). $^{14}C$ atoms are oxidised by carbon-dioxide ($CO_2$) and become mixed throughout the atmosphere and absorbed by oceans and living organisms. It is accepted in general terms that all living organisms absorb $CO_2$ (during tissue-building) in a ratio that is roughly in equilibrium (i.e. will be in similar isotopic ratio) with atmospheric $CO_2$. When the living matter dies, the $^{14}C$ within the organic tissues continue to decay but no replacement takes place and thus if the rate of decay of $^{14}C$ is known then the date of death can be calculated from the measured residual $^{14}C$ activity (with a half life of 5570 + 30 years) (Lowe and Walker, 2014). AMS carbon-14 dating is used to measure the residual $^{14}C$ activity in a sample. To do this the actual numbers of $^{14}C$ atoms in a sample of material are counted by using particle accelerators as mass spectrometers. Accelerator mass spectrometry involves the deflection (by a factor that is proportionate to atomic weight) of charged particles moving in a magnetic field such that the lighter the particle, the greater the amount of deflection.

As the level of atmospheric $^{14}C$ has not been constant for the duration of time that can be radiocarbon dated, a BP date cannot be used as a direct calendar date and thus calibration needs to take place. Reservoirs of carbon also exist in organic matter, the ocean, ocean sediments and sedimentary rocks and changes in the Earth's climate can affect the carbon flows between these reservoirs and the atmosphere, leading to changes in the atmospheric $^{14}C$. In addition to the changes caused by natural processes, the level of atmospheric $^{14}C$ has also been affected by human activities. The fractional level of $^{14}C$ decreased during the industrial revolution (1800s to 1950s), because of the large quantities of $CO_2$ released into the atmosphere because of excavated oil reserves and combustion production of fossil fuel. This decline in atmospheric $^{14}C$ is known as the Suess effect. However, atmospheric $^{14}C$ nearly doubled during the 1950s and 1960s because of atmospheric atomic bomb tests. Thus, the time range covered by radiocarbon dating is >50 000 years and with the exception of using the ‘bomb peak’ it is generally unreliable when dating anything with relatively new carbon (i.e. younger than the 18$^{th}$ century) because of the release of ancient carbon from fossil fuels.

#### *10.3.1.2 Lead-210 ($^{210}$Pb) dating*

Lead-210 ($^{210}$Pb) (an unstable isotope) forms one of the daughter nuclides in a series of nuclides that form part of the Uranium-series decay chain involved in the decay of inert Radon gas ($^{222}$Rn). $^{210}$Pb is removed from the atmosphere as it accumulates in marine sediments, soils, peats and glacial ice and consequently decays over a period of about 150 years to form the stable isotope $^{206}$Pb. As this half-life is relatively short, this technique is used to date soils that are younger than 150 years old (Bennion and Appleby, 1999). If the atmospheric flux of $^{210}$Pb has remained constant, the time that has passed since the lead has been deposited can be determined by measuring the ratio of $^{210}$Pb to $^{206}$Pb in a column of sediment in relation to depth. In this manner, the rate of sedimentation can be ascertained. The main setback of this method is what is termed ‘supported’ $^{210}$Pb and this is due to most sediment containing small amounts of $^{210}$Pb derived from the decay of uranium or its daughters. It is important to determine the ‘supported’ $^{210}$Pb and subtract it from the ‘unsupported’ $^{210}$Pb produced in the atmosphere. The constant rate of supply (CRS) model is used to calculate the inputs of supported and unsupported $^{210}$Pb to the sediment (Appleby, 2001; Appleby et al., 1979). The CRS model assumes that unsupported $^{210}$Pb is being supplied to the sediments at a constant rate. However, this model is convenient, as the rate of sediment accumulation does not have to be constant over time to calculate the age of the sediment accurately. This is useful as recent sediments often have an increasing rate of accumulation owing to erosion caused by modern land-use (as evident at the lowland agricultural site Rhenostervlei Farm). Furthermore, the data were analysed using a second statistical model known as the Slope Regression model to compare results and to be confident that the sedimentation accumulation rates and ages were accurate estimates for the RV3 core. Once the lowest $^{210}$Pb Total Activity (DPM/g) was determined, the Constant Rate of Supply (CRS) sediment accumulation rate was calculated. This was calculated by dividing the cumulative dry mass at the bottom of the extrapolated sediment section by the calculated age at that depth.

The analysis of $^{210}$Pb requires a continuum of sediment to be sampled with each section of sediment amounting to a minimum of 2 g of wet weight. The continuum of sediment should be a maximum of 150 years old or younger (Bennion and Appleby, 1999). Where the depth of the age horizon was not known, samples were systematically taken from the top of the core downwards at predetermined intervals. Samples for $^{210}$Pb dating from the RV3 core were placed in ziplock plastic bags and sent to Core Scientific International in Canada. Here they could equilibrate for three weeks before being measured using an OrtecOctect Alpha spectrometer. This method determines $^{210}$Pb in sediment (dry weights 0.05-0.5g) via the granddaughter (the third chain of the decay chain) Polonium-210 ($^{210}$Po). Using a simple distillation apparatus, the $^{210}$Po is distilled and converted to chloride (Cl-) at 500°C.

The $^{210}Po$ distillate is digested in a nitric acid ($HNO_3$) medium, converted back to the Cl- salt, plated out onto silver (Ag), and then counted by alpha spectroscopy using the OrtecOctect Alpha spectrometer. The recovery is monitored by concurrently measuring the activity of a $^{209}Po$ spike which was added at the beginning of the sample processing (Appleby et al., 1979; Cornett et al., 1994; Eakins and Morrison, 1978).

### 10.3.2 Details on pollen, spores and charcoal extraction from sediment cores

The sample together with distilled water was transferred into 50 ml Nalgene® Centrifuge tubes with sealable caps and stored in a refrigerator. Two exotic spore tablets (*Lycopodium* – Appendix 10.3.6) were added to each 1 $cm^3$ of sediment (Step 1) so that pollen, spore and micro-charcoal concentration and changes in sedimentation rate could be calculated (Bennett and Willis, 2002; Stockmarr, 1971). The *Lycopodium* tablets were produced and distributed by the Department of Quaternary Geology at the University of Lund: batch number 483216 (18583 concentration for one tablet) and batch number 3862 (9666 concentration for one tablet) (see Appendix 10.3.6). The extraction of pollen, spores and charcoal included steps that sequentially removed carbonates (Step 2), humic acids (Step 3), silicates (Step 5) and cellulose using acetolysis (or acetylation; Step 6, Figure 4.1). Samples were mixed using a 'vortex mixer' during each step. In some instances, 10% sodium pyrophosphate ($Na_4P_2O_7$), was used to deflocculate clay before sieving (150 microns) to separate out macro-fossils and large sand and gravel particles (Step 4). Sometimes a second treatment of HF was required to remove all silicates (Step 5), especially in the case of the RV3 core which contained more clay (As) and silt (Ag) – see sediment composition results in section 6.2.

The samples were stained using 2-5 drops of undiluted Safranin solution (Step 7) which increases the contrast of sculptural elements on the pollen grains and spores for easier identification. The samples were adequately dehydrated using tertiary butyl alcohol (TBA) to avoid irreversible clumping when mixing the samples with the mounting medium. Silicone oil was chosen as the mounting medium because it does not set which makes it possible to move and rotate pollen grains by applying gentle pressure on the cover slip during pollen identification (Step 8). At this stage of the extraction procedure, the remaining mixture consisted of silicone oil, stained pollen and spores and micro-charcoal pieces. This mixture was stirred well to ensure that the micro-fossil pollen, spores and charcoal (palynomorphs) were equally distributed throughout the mixture before mounting on glass slides. Glass cover slips were sealed with clear nail polish to keep the samples in place on the slides.

#### 10.3.3 Details of pollen analysis

The number of slides needed to attain these pollen sums varied between one and three. Whilst systematically traversing the slide, pollen grains were counted and tallied using a Java computer programme. Images were captured using a Leica ICC50 HD camera and Leica LAS EZ (V 2.1.0) software and digitally stored. Pollen was identified using a reference slide collection kept in the Department of Biological Sciences, UCT, general pollen reference books (Bonnefille and Riollet, 1980; Moore et al., 1991) and the African Pollen Database (http://medias3.mediasfrance.org/pollen/). Consultation with various palynologists, as well as plant species lists specific to the upland and lowland sites of the focal study area (Manning and Goldblatt, 2012; Mucina and Rutherford, 2006), further facilitated pollen identification. Although the representation of pollen percentage data in palaeoecology is common (Meadows and Baxter, 2001; Meadows et al., 2010; Neumann et al., 2011; Quick et al., 2011; Sugita, 1994; Webb et al., 1981), percentage data may show evidence of the Fagerlind effect, whereby changes in the abundance of one taxon will affect all of the others (Fagerlind, 1952; Prentice, 1988). For this reason, concentration data was used alongside the percentage data to identify any false trends caused by the Fagerlind effect.

Given that Restionaceae and Poaceae pollen grains can look similar, there was uncertainty in identification of Restionaceae versus Poaceae in the initial phases of pollen counting (i.e. first 8-16 samples). Although the uncertainty in identifying these pollen grains decreased over time, to increase our confidence, after the count was completed for each core (38 levels for RV3 and 53 levels for GWWA006) an aliquot of 4 to 8 levels was re-counted for Restionaceae and Poaceae only to confirm identification.

Selected pollen taxa, particularly Asteraceae *Stoebe/Elytropappus*-type and Asteraceae high spine type-1, were disaggregated by size classes to further interrogate the data for regional versus local signals. The smaller size class is a proxy for a stronger regional signal and the larger size class is a proxy for a local signal.

#### 10.3.4 Details of spores analysis

Coprophilous fungal spores are spores associated with fungi that grow on animal dung (Davis and Shafer, 2006; Graf and Chmura, 2006), decaying wood (Lundqvist, 1972; Massee and Salmon, 1901) and/or associated with human dwelling sites (van Geel et al., 2003). Rather than relying solely on one taxa, the widely accepted coprophilous fungal spore *Sporormiella*, a range of coprophilous fungal spore types were analysed to track herbivory/grazing over time, namely *Sporormiella*, Sordariaceae, *Gelasinospora* and *Coniochaeta* (Baker et al., 2013; Carrión et al., 2000; Forbes et al., 2018; Gelorini et al., 2011; Graf and Chmura, 2006; van Geel and Aptroot, 2006). The same slides used for pollen identification were used for the identification of coprophilous fungal spores. In a similar way to the pollen counts, the calibrated stage of the microscope was used to traverse the slide systematically at 1 mm intervals to count spores.

#### 10.3.5 Details of charcoal analysis

The point count estimation method used in this study allowed for standardization, therefore charcoal concentrations could be compared at different levels (Clark, 1982). A sampling grid was created to traverse the slide and sample all areas of the slide equally. The eyepiece graticule was used to sample the field of view (f.o.v.). The points form the ends of the 11 lines of the eyepiece graticule (major units) resulting in a total of 22 sampling points. The same slides that were used for pollen and spore analysis were used for the assessment of micro-charcoal abundance. Assessment was done at 400x total magnification and a minimum of 200 items (sum of charcoal hits and *Lycopodium* spores) were counted (Finsinger and Tinner, 2005), and at least 100 charcoal particles were counted in each sample (Duffin et al., 2008). Compared to pieces of organic material, charcoal was identified by its dark colour and its characteristic sharp angular edges. Only charcoal pieces greater than 10 µm in length were recorded and any ambiguous fragments were excluded (Mooney and Tinner, 2011). Micro-charcoal concentration was calculated by using an equation modified from Bennett and Willis (2002), with the surface area of charcoal per unit volume of sediment expressed in $cm^2\ cm^{-3}$:

At each 1 mm point on the sampling grid the f.o.v. of the microscope was examined and according to standard procedures (Clark, 1982), the micro-charcoal abundance was assessed by recording when the points (the end of the lines of the eyepiece graticule) "touched" a piece of charcoal. The total number of *Lycopodium* spores in each f.o.v. was also recorded. The equation modified from Bennett and Willis (2002) to calculate micro-charcoal concentration ($cm^2\ cm^{-3}$) includes the following:

- 'f.o.v.' is the area of the field of view in microns;
- 'no. of charcoal hits' is the total number of points that coincided with a charcoal fragment;
- 'no. points' is the largest number of possible points that can coincided with a charcoal fragment.
- '*Lycopodium* added' is the number of *Lycopodium* spores contained in the tablets added to the 1 $cm^3$ sample taken at each depth (i.e. 18583 x 2 or 9666 x 2, depending on which samples were analysed (see Appendix 10.3.6)).

- '*Lycopodium* counted' is the number of *Lycopodium* spores counted at each depth; and
- 'sediment volume' is the volume of sediment analysed in the laboratory for each depth (i.e. 0.5-2 $cm^3$).

Macro-charcoal (>150 µm) was recovered during the extraction procedures described in Step 4 (Figure 4.1) in section 4.1.3.2 above, and stored in distilled water before bring poured into a petri dish for examination. When there were few charcoal particles, all were counted. Where charcoal was extremely abundant (>100 pieces per level), a sub-sampling exercise was performed where only a proportion of the petri dish was counted, then the count was multiplied to obtain the total charcoal fragments.

### 10.3.6 Information on Lycopodium spore tablets used for pollen, spore and charcoal extraction and analysis

Spore tablets for calibration of pollen analyses have earlier been produced and distributed by Dr Jens Stockmarr, Copenhagen. In October 1980 this business was taken over by the Department of Quaternary Geology in Lund. It is performed as an official commission approved by the University of Lund. A batch from September 2004 (No. 483216) was produced, calibrated and tablets were used in the initial sub-sample pollen analysis for the RV3 core. These tablets were manufactured in Denmark. A new batch, No. 3862, is now produced and tablets were used for the remaining sub-sampling of the RV3 core and GWWA cores. These tablets were prepared, pressed and calibrated in Sweden.

**Lycopodium spore tablets (batch 483216) (September 2004):**

*Lycopodium* spore tablets can be dissolved in water or in HC1, but not in NaOH. For this study the *Lycopodium* spore tablets were prepared in a slightly different way compared to that described by Stockmarr (1971, 1973). The tablets comprises mainly of sodium bicarbonate together with polyvinylpyrilidon and polyethyleneglycol, thus they had to be carefully washed away with water and finally with diluted HCl before further treatment. The spores were acetolysed. The spore concentration was determined with an electronic particle counter, Coulter Counter ZB (cf.Stockmarr 1973) with a tube size of 140 pm. Preparation included 100 samples of five tablets each that were taken from different places in the batch. These samples were prepared by dissolving the tablets in Isoton II NaCl solution in 100 ml flasks, with 20 counts each of 0.5 ml made on each sample.

Results of the calibration for 5 tablets showed the following concentration:

X = 92914; sd = ± 3820; V = ± 4.1 %. Thus the Lycopodium spore concentration for one tablet is: X = 18583.

**Lycopodium spore tablets Batch 3862 (June 2014)**

*Lycopodium* spore tablets can be dissolved in water or in HCl, but not in NaOH. For this study the *Lycopodium* spore tablets have been prepared in a slightly different way compared to that described by Stockmarr (1971, 1973). The tablets comprises mainly of sodium carbonate together with polyethyleneglycol, thus they had to be carefully washed away with water and finally with diluted HCl before further treatment. New for this batch is that the sodium bicarbonate has been replaced by sodium carbonate. The spores were acetolysed. The spore concentration was determined with an electronic particle counter, Coulter Counter ZB (cf. Stockmarr 1973), tube size 100 µm. Preparation included 100 samples of ten tablets that were taken from different places in the batch. These samples were prepared by dissolving the tablets in Isoton II NaCl solution in 100 ml flasks with 5 counts each of 1 ml made on each sample. Result of the calibration for 10 tablets showed the following concentration: X = 96 660 s = ± 2123 V = ± 2.2 %. Thus, the Lycopodium spore concentration for one tablet is: X = 9666.

### 10.3.7 Details of diatom analysis

Systematic steps for diatom extraction were adapted from the general procedure for extracting diatoms from sediments by Battarbee (1986). Laboratory work was conducted in the Palaeoecology Lab in the Department of Biological Sciences, UCT. Using a plastic 3 ml medical syringe with the tip removed, sub-samples were taken at 2-4 cm intervals along the 108 cm core. Samples were treated with 20 ml of 30% $H_2O_2$ to break down organic material while being heated gently in an 80 $^0$C water bath for approximately 3 hours (Step 1, Figure 4.2). $H_2O_2$ was added in 5 ml fractions and ethanol was used to stop samples from bubbling out of the test tubes. Samples were swirled intermittently over the duration of the reaction to ensure all organic matter had been removed. Once the reaction was complete, distilled water was added to the 45 ml mark on the tubes and was left at room temperature overnight. The supernatant was decanted while ensuring no sediment at the bottom of the test tube was lost.

To remove all carbonates from the sediment, 10 ml of 10% HCl was added to each sample whilst in a 80 $^0$C water bath and test tubes were swirled intermittently (Step 2, Figure 4.2). If the carbonates were not fully digested, an additional 10 ml of HCl was added. The reaction was complete when the liquid within the test tubes turned a luminous yellow colour and was close to transparent. Distilled water was added to each sample to the 45 ml mark and was left overnight. The supernatant was decanted and the sample was sieved through a 0.25 mm (Step 2, Figure 4.2) to remove coarse material, including plant, organic matter and sediments. Samples were decanted into 100 ml beakers using as minimal distilled water as possible to clean out the test tubes. Samples were swirled for a few seconds before being decanted back into the test tubes while leaving the coarse mineral matter in the beaker. Additional distilled water was added to the beaker and the swirling (Step 4, Figure 4.2) and decanting steps were repeated until the test tube was filled to 45 ml. To remove the finer clay particles, the samples were washed by subsequent refilling and decanting of distilled water (Step 5, Figure 4.2). Once samples were filled to the 45 ml mark, they were settled for 8 hours and then the supernatant was decanted with the fine clay particles in suspension being removed with it. This step was repeated (between 10-15 times) until the supernatant appeared to be clear. Prepared diatom samples (approximately 0.2 ml) were mounted on glass slides using rectangular glass cover slips and Pleurax (R.I. = 1.73) (Step 6, Figure 4.2). Pleurax creates a dry mount as it sets which makes it impossible to move the diatoms when applying pressure on the cover slip during diatom identification but allows for optimal visualisation of the diatom valves under the microscope.

### 10.3.8 Details on Magnetics

Sample preparation procedure for Xlf, ARM, IRM are as follows:

1. Sediment samples were dried at 40 degree Celcius in an oven. Depending on the moisture content of samples, this step took between 1-2 days.
2. Each sample was ground using a mortar and pestle to separate soil/sediment particles. In order not to break the particles by applying too much pressure, samples were ground up lightly.
3. Each ground sample was mixed thoroughly in the mortar to ensure even distribution of the sample.
4. The sample was sieved using a 0.063 mm sieve used for both magnetics and LOI measurements.

Furthermore, the same samples that were ground using a pestle and mortar and processed for XRF, were measured for volume MS (reported in units SI) using the *MS3 Magnetic Susceptibility Meter*. They included 86 sub-samples (49 for GWWA006 and 37 for RV3). The following steps were followed:

1. The ground up sediment samples were placed into a 2 ml plastic vial. Given the limitation of the number of vials available, the samples were processed in batches of 10 with three replicates measured for each sub-sample depth to eliminate error.

2. After the samples were measured, they were decanted back into plastic pots for storage and the 2 ml vials were thoroughly cleaned using compressed air to remove traces of sample before processing the next batch of samples.
3. MS3 meter is designed to work with proprietary Bartington Instruments Bartsoft software on a Windows computer from which the volume MS data was downloaded to Excel for further analysis.

### 10.3.9 Details on age-depth modelling

This SHCal20 calibration dataset is a combined set of dendrochronologically-dated records from sites in the Southern Hemisphere and can be confidently applied to $^{14}C$ measurements to account for the temporal difference between hemispheres (Hogg et al., 2020). Based on the ages generated by the age-depth model, average sediment accumulation rates were calculated (cm/year) according to the zones determined by the cluster analysis described below. Modelled calibrated ages were plotted next to the pollen, spore and charcoal data using R and then plotted using C2.

*R script for age-depth modelling - example from RV3 dataset*

```
Bacon("RV3.dates", thick = 2, mem.mean = 0.4, d.max = 108, BCAD = TRUE,
rev.age=TRUE, rev.yr = TRUE, title = "RV3", coredir = "chronology_practical/")
```

### 10.3.10 Details on statistical analysis of palaeoenvironmental data

#### *10.3.10.1 Cluster analysis (zonation)*

The number of statistically significant clusters in the pollen and diatom diagrams were identified using constrained hierarchical clustering in the packages '*rioja*' (Juggins, 2015) and '*vegan*' (Oksanen et al., 2013) for R (R Core Team, 2017). Since the pollen and diatoms datasets represent species abundance data, the percentage data was square root transformed and the concentration data was transformed using the Hellinger method, which divides each value in a data matrix by its row sum, and taking the square root of the quotient to best represent ecological relationships while not violating statistical procedure (e.g. normally distributed values). The transformation gives low weights to variables with low counts and many zeros. Thereafter, a distance matrix containing all pollen and diatom types was constructed using the Bray-Curtis method, and a clustering process was then performed on this matrix using the 'CONISS' method of agglomeration with clusters constrained by sample order (in this case depths) (Bennett, 1996; Birks and Gordon, 1989). The number of statistically significant clusters (zones) was determined by comparing the spread of the cluster analysis with that of random simulations in a scree plot using the broken-stick method (Oksanen et al., 2013). The eigenvalues expected under the curve of the broken stick model represented non-significant components (Jackson 1993). Given the objectives of the present study, sub-zones that showed subtle changes in the palaeo-record were identified using the cluster analysis although they were not statically significant. The zonation results were presented on the pollen and diatom diagrams using C2.

*R script for percentage pollen zonation - example from RV3 dataset*

```
RV3.pollen.depth <- read.csv("/R data analysis/RV3.pollen.perc.SAUL.csv", header = TRUE)
RV3.depth.mat <- RV3.pollen.depth[, "depth"]
yticks <- seq(0, max(RV3.depth.mat), by = 5)
RV3.pollen <- select(RV3.pollen.depth, -depth)
```

#If percentage data then transform the % pollen data then do the cluster

```
RV3.pollen.transform <- dist(sqrt(RV3.pollen))
```

#If concentration data then transform using Hellinger and create a distance matrix

```
RV3.pollen.transform.conc <- decostand (RV3.pollen.conc, method = "hellinger")
RV3.pollen.transform.conc.dist<-dist(RV3.pollen.transform.conc)
RV3.cluster.constrained <- chclust(RV3.pollen.transform, method = "coniss")
jpeg("Sum of squares_broken stick_perc.jpeg")
bstick(RV3.cluster.constrained)
dev.off()
jpeg("Zonation.perc.jpeg")
RV3.stratigraphy <- strat.plot(RV3.pollen, yvar = RV3.pollen.depth$depth,
scale.percent=TRUE, y.rev = TRUE, y.tks = yticks, srt.xlabel = 45, yTop = 0.7, xRight =
0.9, xLeft = 0.15, ylabel="Depth (cm)", x.names=NULL, cex.axis=0.8, clust =
RV3.cluster.constrained)
addClustZone(RV3.stratigraphy,clust = RV3.cluster.constrained, 4, col = "red")
dev.off()
```

#### *10.3.10.2 Non-metric multidimensional scaling (NMDS) ordination*

For the non-metric multidimensional scaling (NMDS) a Bray-Curtis Similarity was used to analyse similarity between samples that reflected the different depths of the cores based on pollen taxa composition. Where data values were larger than the common class scales, a Wisconsin double standardisation and square-root transformation were performed. Several model runs (at least 30) were performed to be confident that a global solution was found. Thereafter, the function called *'envfit'* was used to fit environmental variables onto the ordination. The similarity matrix of the pollen taxa and depths were fixed, while subsets of the "environmental variables" (e.g. herbivory, micro- and macro-charcoal, magnetic susceptibility, etc.) were used in the calculation of the environmental similarity matrix. A correlation coefficient was calculated between the two matrices and the best subset of environmental variables was further subjected to a permutation test to determine significance. The NMDS ordination of pollen taxa and depths was then plotted with best correlating environmental variables as vectors.

*R script NMDS - example from RV3 dataset*

```
library(vegan)
library(tidyverse)
library(ggrepel)
library(dplyr)
library(viridis)
data <- readxl::read_excel("/R data analysis/RV3_pollen_data.xlsx") %>%
  select(-depth)
```

#Read in the environmental data like diatoms, spores, charcoal, XRF, MS

```
env <- readxl::read_excel("/R data analysis/RV3_env_data.xlsx")
```

#Calculate Bray-curtis distances

```
bcdist<-vegdist(data,method="bray")
scree_values1<-function(data,max,dist_meas)
{
  xx<-as.matrix(data)
  scree.points=NULL
  scree.temp=1
  for(i in 1:max)
  {
    sol.mds=NULL
    sol.mds<-replicate(10,metaMDS(xx,k=i,trymax=30,distance=dist_meas)$stress,
                       simplify=TRUE)
    scree.points<-append(scree.points,sol.mds)
  }
  return(scree.points)
}

scree.res1<-scree_values1(data,max=10,dist_meas="bray")
par(cex=1.5,las=1)
plot_vec1<-
as.vector(sapply(1:(length(scree.res1)/10),function(x)rep(x,10),simplify="vector"))
```

#Prerequisite for plotting replicated run of mds

```
jpeg("Stress_values.jpeg")
plot(plot_vec1,scree.res1,ylab="Stress1",xlab="dimensions")
lines(1:(length(scree.res1)/10),as.vector((by(scree.res1,plot_vec1,mean))))
abline(h=0.10,col="red")
dev.off()
```

#The red line indicates threshold for good Stress value and therefore the number of k dimensions to use. Run the MDS

```
mds1<-metaMDS(data, distance="bray",trace=FALSE, k=4)
mds1
jpeg("Stress_plot.jpeg")
stressplot(mds1)
dev.off()
sites1 <- as.data.frame(scores(mds1))
sites1$site <- rownames(sites1)
sites1$grp <- env$Zones
head(sites1)
glimpse(sites1)
species1 <- as.data.frame(scores(mds1, "species"))
species1$species <- rownames(species1)
head(species1)
```

#Run Envfit on the species scores

```
set.seed(123)
ef1 <- envfit(mds1, env[,c(7,12,13,14,15,18,24,26)], perm = 999)
summary(ef1)
jpeg("Preliminary NMDS_plot.jpeg")
scores(ef1, "vectors")       # or "factors"
plot(mds1)
plot(ef1, col = "black")
dev.off()

ef.scrs1 <- as.data.frame(scores(ef1, display = "vectors"))
head(ef.scrs1)
ef.scrs1 <- cbind(ef.scrs1, Sites = rownames(ef.scrs1))
head(ef.scrs1)
```

#Plot the diagram

```
ggplot() +
```

```
  geom_point(data = sites1, aes(x = NMDS1, y = NMDS2, colour = grp),size=2,shape = 15) +
##size=2 is the size of the dots of the sites/depths
  scale_color_manual(values = c("blue","green"),
                     name = "Zones",
                     labels = c("R1","R2"))+
  geom_point(data = species1, aes(x = NMDS1, y = NMDS2),size=1) +
  coord_fixed(ratio = 1, xlim = c(-0.85,0.9), ylim = c(-0.5,0.9), expand = TRUE) +
  geom_segment(data = ef.scrs1, aes(x = 0, xend = NMDS1, y = 0, yend = NMDS2),
               arrow = arrow(length = unit(0.2, "cm")), colour = "#969696") +
geom_text_repel(data = ef.scrs1, aes(x = NMDS1, y = NMDS2, label = Sites),size = 5, col =
"red")+
  geom_text_repel(data = species1, aes(x = NMDS1, y = NMDS2, label = species),size = 4)+
  labs(x="Axis 1", y="Axis 2") +
  theme_bw()+
  theme(axis.text = element_text(size = 13, color = "black"),
        legend.text = element_text(size = 15, color = "black"),
        legend.title = element_text(size = 15, color = "black"),
        axis.title = element_text(size = 15, color = "black"))

ggsave("NMDS_species_FINAL_new.jpeg", width = 15, height = 8)
ggsave("NMDS_species_FINAL_new.pdf", width = 15, height = 8)
```

## 10.4 Additional details on PSD methods

### 10.4.1 Factors that influenced the modelling Validation Cessation Threshold (VCT)

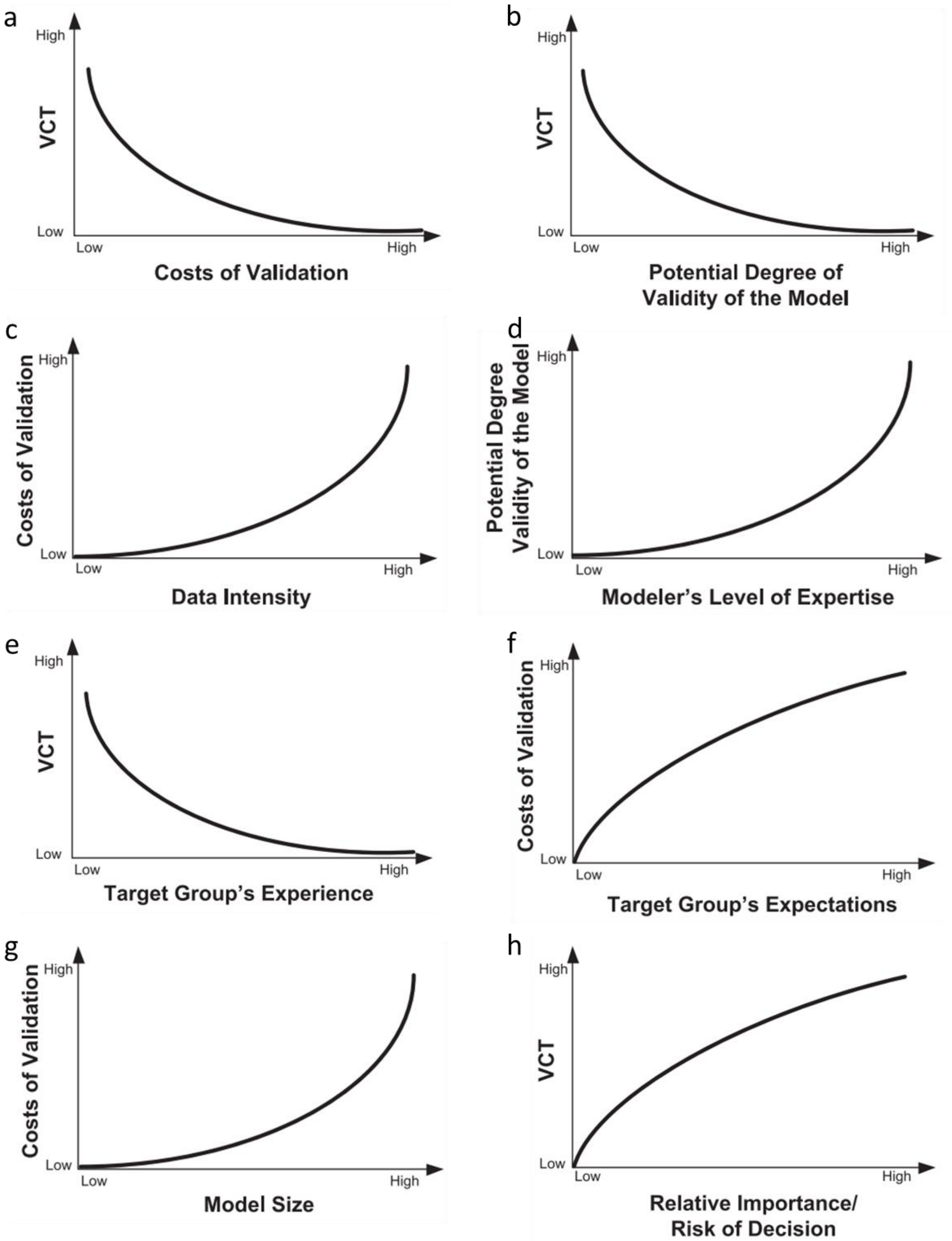



*Figure 10.1: Graphs showing how factors influence the validation cessation threshold (VCT) (a, b, e and h); costs of validation (c, f and g); and potential degree of validity of the model (d); adapted from Groesser and Schwaninger (2012).*

With regards to graph d in Figure 10.1, the following was done to increase my level of SD expertise and therefore lower the validation cessation threshold (VCT). To increase my modelling experience in preparation for the multi-stakeholder engagement workshop I conducted a mock workshop "*Testing an approach: Using a system dynamics methodology to investigate changes in ecosystem services over time"* (14 May 2018) with researchers within the Department of Biological Sciences and African Climate and Development Initiative, UCT. I received training in system dynamics during the *Introduction to System Dynamics Modelling* short course (28 May-8 June 2018) facilitated by the School of Public Leadership, Stellenbosch University. I received training and experience as a table host in the following workshops: (1) *Transforming the Nature Conservation and Game Ranch Management programs to be more Africanised and globally competitive* (23 October 2018, Nelson Mandela University, George Campus); and (2) *Embedding multi-level learning processes within in the GEF 5 governance network* (6 November 2018, Mountain Zebra Camdeboo Protected Environment, Eastern Cape), which was a part of research on adaptive capacity in resource governance. Further capacity development was required so I participated in a six-week *System Dynamics Modelling Process* course (November-December 2019, University of Bergen, Norway).

#### 10.4.2 Appendix: Semi-structured interviews with key stakeholders

These templates were used to capture relevant SES contextual information and guide discussions with key stakeholders regarding land-use characteristics related to ecosystem services and drivers. Responses were captured electronically and therefore space for in-depth responses are not evident in these templates.

<table>
<tr><th colspan="4">SUMMARY INFORMATION FOR [insert case study coring site name]</th></tr>
<tr><td>Date of veg survey:<br><br>Date of coring:</td><td>Location name:</td><td>Site name and no.:</td><td rowspan="2">Extra notes on coring:</td></tr>
<tr><td>Weather:</td><td>Elevation:</td><td>Co-ordinates:</td></tr>
<tr><td colspan="2"></td><td colspan="2">Google map images and photos of site and location of core</td></tr>
<tr><td>Sediment depth (testing):</td><td>Basin size:</td><td colspan="2" rowspan="5"></td></tr>
<tr><td>Sediment texture &colour:</td><td>Disturbance level:</td></tr>
<tr><td>Depth of water:</td><td rowspan="2">Short list of vegetation near coring site:</td></tr>
<tr><td>Source of water at site:</td></tr>
<tr><td>Depth of core:<br><br>Originally when cored:<br>Total depth analysed:</td><td>Photo no. for important photos:</td></tr>
</table>

## SOCIAL-ECOLOGICAL SYSTEM (SES) CONTEXT

**Superficial description of the major communities (veg units) surrounding the coring site. How many, major components, etc.?**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**General vegetation pattern (see google earth map) - comment on landscape patchiness/heterogeneity:**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**Size of wetland and pollen source area (see Table: Relationship between the size of the site cored and the pollen source area):**

**Relations of the vegetation types with...**

**Climatic artefacts (fog, heavy rainfall, flooding, drought, etc.):**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**Geology and soil conditions:**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**Topography (rocky outcrops as fire refugia):**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**Fire scars:**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

**Erosion (how water flows):**

- Vegetation unit 1:
- Vegetation unit 2:
- Vegetation unit 3:
- Etc.

***Table 10.1: Summary of information related to research participants during the course of the research. The form of contact is defined: SI = semi-structured interview; E = email discussion; D = documents provided; R = regular discussions; Workshop = W; 'Key informants' are shaded grey. Land-use sector representation: CF = Commercial Farming; CP = Conservation Practitioner; GA = Government Authority.***

| Stakeholder group | Land-use sector representation | Organisation and position | Form of contact | Dates |
|---|---|---|---|---|
| Primary | CF:1 | Rhenostervlei Farm, Farmer and Landowner | I; E; D; R | 2011; 2017-2021 |
| | CF:2 | Suurvlakte Farm, Farm Manager | I | 10/2020 |
| | CF:3 | Suurvlakte Farm, Farm Manager | I | 10/2020 |
| | CF:4 | Agtergroenberg Renosterveld Conservancy, Elandsberg PNR and Bartholomeus Klip, Office Manager | E | 07/2019 |
| | CF:5 | Agtergroenberg Renosterveld Conservancy, Bontebok Ridge Reserve and Limietrivier Farm, Farm Manager | E | 07/2019 |
| | CF:6 | Farmer | E | 07/2019 |
| | CF:7 | Simonsig Wine Farm, Marketing Manager | I; R; W | 2016-2021 |
| | CP:1 | Elandsberg PNR, Nature Reserve Manager | I; E; D; R | 2012-2014; 2019-2021 |
| | CP:2 | CapeNature, Regional Conservation Intelligence Manager | I; E; D; R; W | 2016-2021 |
| | CP:3 | CapeNature, Conservation Manager for Rocherpan Nature Reserve | W | 07/2019 |
| | CP:4 | CapeNature, Ecological Technician | W | 07/2019 |
| | CP:5 | CapeNature, GWWA Reserve Manager | E | 07/2019 |
| | CP:6 | CapeNature, GWWA Acting Reserve Manager | I; E; D | 10/2016 |
| | CP:7 | CapeNature, GWWA Field Ranger | I; E; D | 2016 |
| | CP:8 | CapeNature, GWWA, Reserve manager | I; E; D; R | 2020-2021 |
| | CP:9 | CapeNature, Field Ranger | I | 10/2020 |
| | CP:10 | SANBI, Research Manager<br>Stellenbisch University, EbA Researcher | I; E | 2016-2019 |
| | CP:11 | SANBI, Senior Scientist | E | 06/2019 |
| Secondary | GA:1 | Western Cape Department of Environmental Affairs and Development Planning, Ecological Infrastructure and Bioremediation Manager | W | 07/2019 |
| | GA:2 | Western Cape Department of Agriculture, LandCare, Chief Industrial Technician | I; E; D; R; W | 2018-2021 |
| | GA:3 | Western Cape Department of Agriculture, Cape Winelands District Manager | I; E; D; R | 2018-2022 |
| | GA:4 | Western Cape Department of Agriculture, LandCare, EngineeringTechnician | W | 07/2019 |
| | GA:5 | National Department of Forestry, Fisheries and the Environment, Director of Carbon Sinks Mitigation | E; R | 2018 |
| | GA:6 | National Department of Forestry, Fisheries and the Environment, Deputy Director of Biodiversity and Climate Change | E; R | 2016-2018 |

### 10.4.3 Details on multi-stakeholder engagement workshop

*Table 10.2: Agenda for multi-stakeholder engagement workshop for variable elicitation. This version includes the table host prompts used by the facilitation team – page 1/2.*

**Stakeholder Engagement Workshop 1: Sustainable land-use management in the Lower Berg River Catchment**

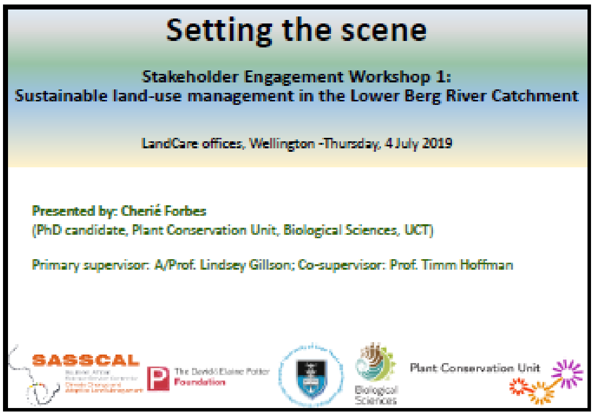
**Date:** Thursday, 4 July 2019; **Time:** 08h30 – 15h30; **Venue:** 2nd Floor, Wellington Centre, Church Street, Wellington

| Duration | Time | Event | Persons involved | Table host PROMPTS |
|---|---|---|---|---|
| 30 mins | 08:30 – 09:00 | Tea and Registration | All participants | |
| 20 mins | 09:00 – 09:20 | Welcome and Introductions | Cherie Dirk & participants | |
| 15 mins | 09h20 – 09h35 | **Setting the scene** - Overview of project & objectives of workshop | Cherie Dirk | |
| 25 mins | 09h35 – 10h00 | ***Dialogue 1: Natural resources (Ecosystem Services (ES))***<br>*a) What natural resources from nature/the land do you use most in the context of your work? **(15 mins)***<br>*b) Why are these three resources (biodiversity, water quality and soil erosion regulation) relevant to you and/or other stakeholders in this catchment? **(10 mins)*** | All participants | a) Natural resources = ES/services we get from nature.<br>- "...in the context of your work as a farmer/conservation practitioner/researcher/manager"<br><br>b) Focus on the particular ES/resource: which stakeholder uses/produces this and why is it important to you/other stakeholders? |
| 25 mins | 10h00 – 10h25 | **Dialogue 1 (2nd iteration)** - *Participants shift tables have the same conversation with a <u>different group of people</u>* | All participants | |
| 25 mins | 10h25 – 10h50 | **Dialogue 1 (3rd iteration)** - *Participants shift tables have the same conversation with a different group of people* | All participants | |
| **30 mins** | **10h50 – 11h20** | **TEA** | | |
| 10 mins | 11h20 – 11h30 | Feedback on Dialogue 1 | Table hosts | Summarise key points |
| 25 mins | 11h30 – 11h55 | ***Dialogue 2: Influencing factors (concerns / threats / drivers of change)***<br>*a) What makes it difficult for you to manage land in the context of your work? **(15 min)***<br>*b) Why are these three influencing factors (fire, herbivory/grazing and climate change) important to you and/or other stakeholders in this catchment? **(10 min)*** | All participants | **a) Influencing factors = concerns / threats / drivers / impacting factor.**<br>- "...in the context of your work as a farmer/conservation practitioner/researcher/manager"<br><br>**b) Focus on the particular** |

*Agenda for multi-stakeholder engagement workshop for variable elicitation – page 2/2*

| | | | | |
|---|---|---|---|---|
| 25 mins | 11h55 – 12h20 | **Dialogue 2 (2nd iteration)** - *Participants shift tables have the same conversation with a different group of people* | All participants | **driver/impacting factor:**<br>- which stakeholder is impacted by this and why is it important to you/other stakeholders?<br>- when there is too much or too little of these impacting factors how does it affect your work (farming/conservation/governance)? |
| 25 mins | 12h20 – 12h45 | **Dialogue 2 (3rd iteration)** - *Participants shift tables have the same conversation with a different group of people* | All participants | |
| | **12h45 – 13h30** | **LUNCH** | | |
| 15 mins | 13h30 – 13h45 | Feedback on Dialogue 2 | Table hosts | Summarise key points |
| 25 mins | 13h45 – 14h10 | ***Dialogue 3: Connections (links / relationships / cause and effect)*** *How do think these natural resources (biodiversity, water quality, soil erosion regulation), impacting factors (fire, herbivory, climate change) and stakeholders relate to each other - where are the connections in the system?* | All participants | Connections = links / relationships / cause and effect<br>- cause = **what happened** for this to happen?<br>- effect = **what happens** because of this?<br>- do you think this is a direct or indirect link?<br>- **mind mapping** may be very useful here! |
| 25 mins | 14h10 – 14h35 | **Dialogue 3 (2nd iteration)** - *Participants shift tables have the same conversation with a different group of people* | All participants | |
| 15 mins | 14h35 – 14h50 | Feedback on Dialogue 3 | Table hosts | Summarise key points |
| 20 mins | 14h50 – 15h10 | Any additional input and concluding remarks | All & lead modeller | |
| 10 mins | 15h10 – 15h20 | Way forward | Cherie Dirk | |
| 11 mins | 15h20 – 15h30 | Complete workshop evaluation form | All participants | |
| | | **CLOSE** | | |

*PowerPoint presentation slide layout – page 1/6*

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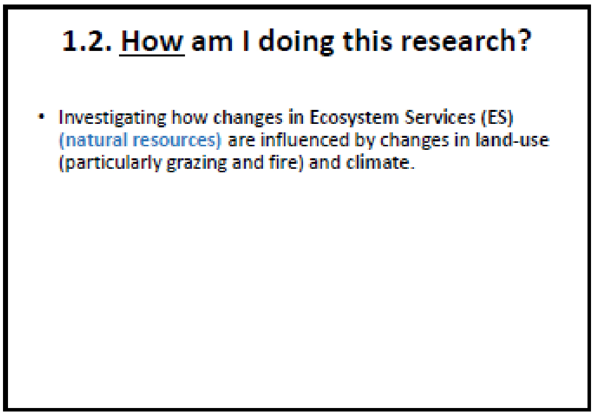


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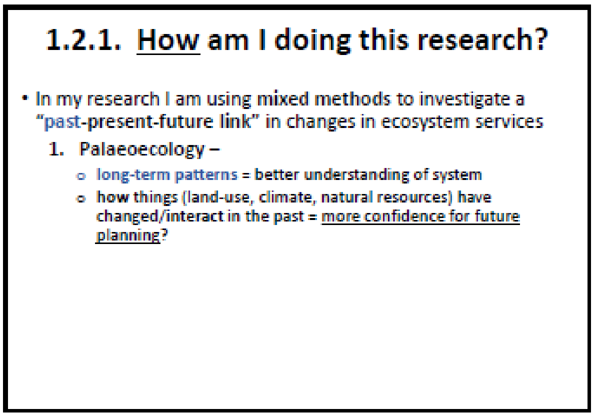


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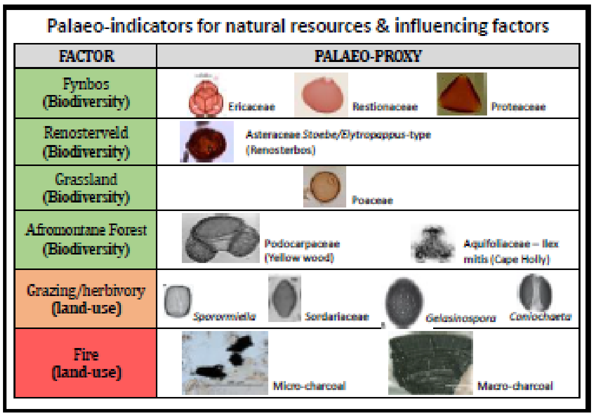


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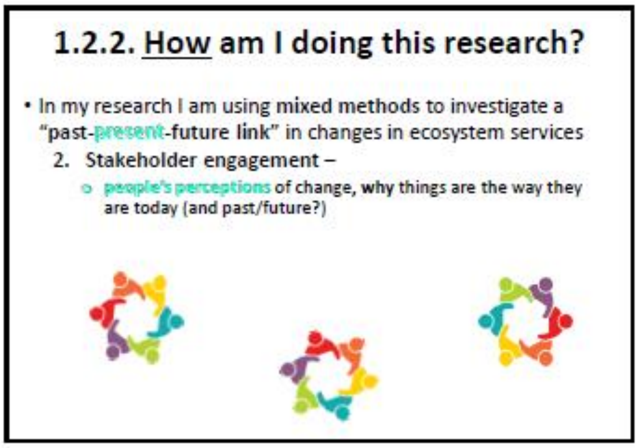
***Figure 10.2: PowerPoint presentation slides presented at the multi-stakeholder multi-stakeholder engagement workshop.***

*PowerPoint presentation slide layout continued – page 2/6*

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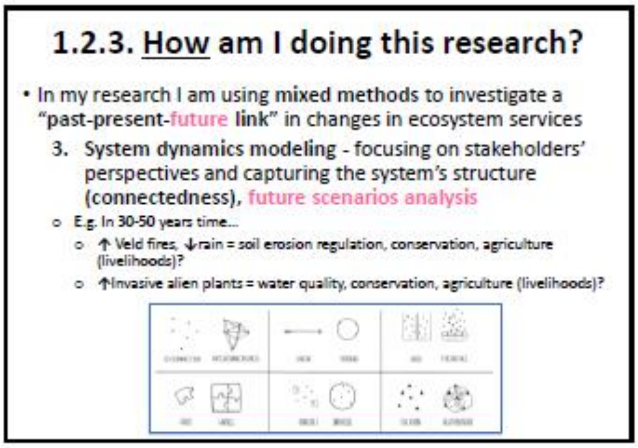


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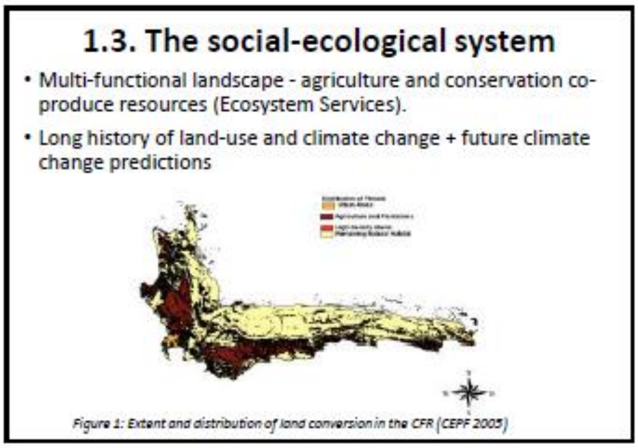


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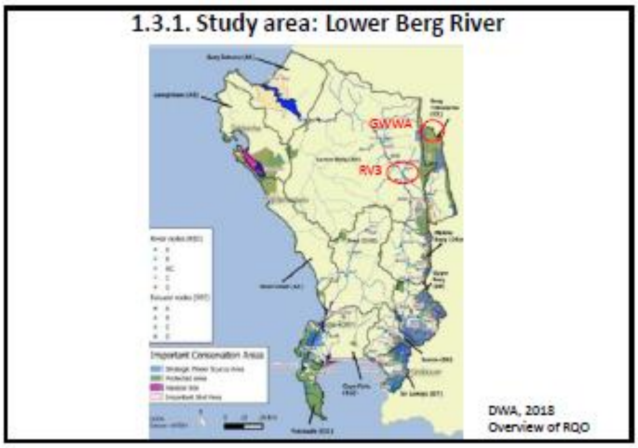


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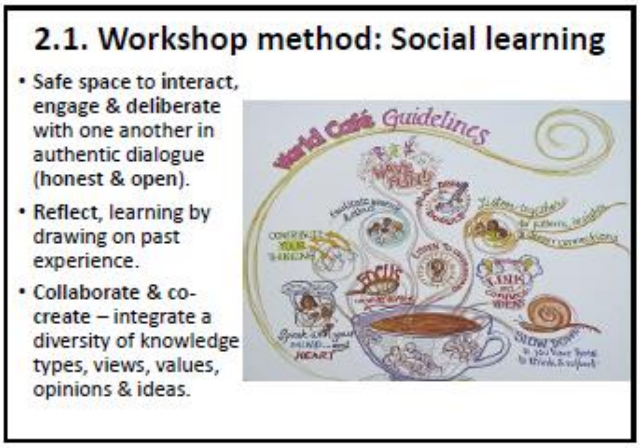


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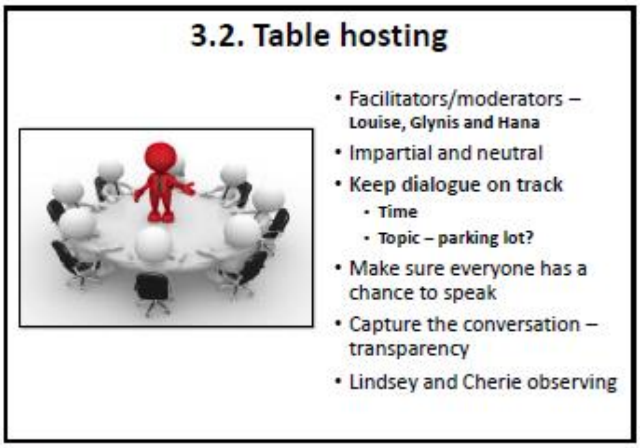


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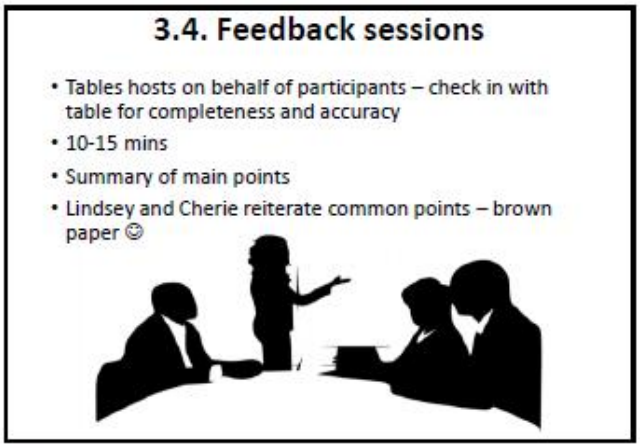


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*PowerPoint presentation slide layout continued – page 4/6*

## 5. Logistics

- Sign register
- Sign ethics form – participate in research, audio-recordings
- Photos
- Tools
  - Sticky notes and pens
  - Parking lot on brown paper
- Workshop evaluation forms – participants feedback on process

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## Agenda: Land-use management Workshop 1 dialogues

| Time | Event | Persons Involved |
|---|---|---|
| 08:30 – 09:00 | Tea and Registration | All participants |
| 09:00 – 09:20 | Welcome and Introductions | Cherie Forbes and all participants |
| 09h20 – 09h35 | Setting the scene - Overview of PhD project & objectives of the workshop | Cherie Forbes |
| 09h35 – 10h50 | ***Dialogue 1: Natural resources (Ecosystem Services (ES))*** <br> ***a) What natural resources from the land do you use most in the context of your work?*** <br> ***b) Why are these three resources (biodiversity, water quality and soil erosion regulation) relevant to you and/or other stakeholders in this catchment?*** | All participants |
| 10h50 – 11h20 | TEA | |
| 11h20 – 11h30 | Feedback on Dialogue 1 | Table hosts |

20

## Agenda: Land-use management Workshop 1 dialogues

| | | |
|---|---|---|
| 11h30 – 12h45 | ***Dialogue 2: Influencing factors (concerns / threats / drivers of change)*** <br> ***a) What makes it difficult for you to manage land in the context of your work?*** <br> ***b) Why are these three influencing factors (fire, herbivory/grazing and climate change) important to you and/or other stakeholders in this catchment?*** | All participants |
| 12h45 – 13h30 | LUNCH | |
| 13h30 – 13h45 | Feedback on Dialogue 2 | Table hosts |
| 13h45 – 14h35 | **Dialogue 3: Connections (links / relationships / cause and effect)** <br> *How do think these natural resources (biodiversity, water quality, soil erosion regulation), influencing factors (fire, herbivory, climate change) and stakeholders relate to each other - where are the connections in the system?* | All participants |
| 14h35 – 14h50 | Feedback on Dialogue 3 | Table hosts |
| 14h50 – 15h10 | Any additional input and concluding remarks | All participants & Benjamin Balinga |
| 15h10 – 15h20 | Way forward | Cherie Forbes |
| 15h20 – 15h30 | Complete workshop evaluation form | All participants |
| | CLOSE | |

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# Start of Dialogues

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## Dialogue 1: Natural resources (ecosystem services)

a) *What natural resources from the land do you use most in the context of your work?*

b) *Why are these three resources (biodiversity, water quality and soil erosion regulation) relevant to you and/or other stakeholders?*

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## Dialogue 1


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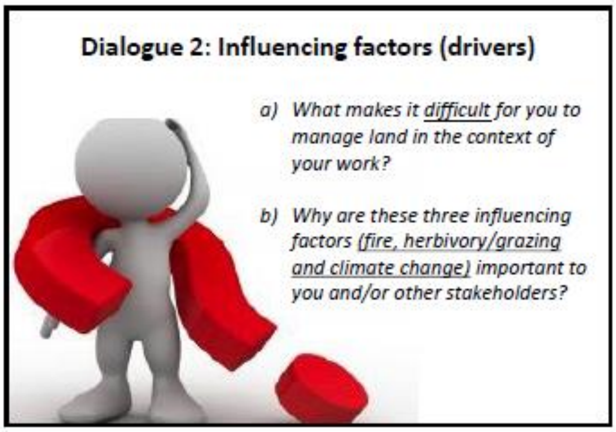


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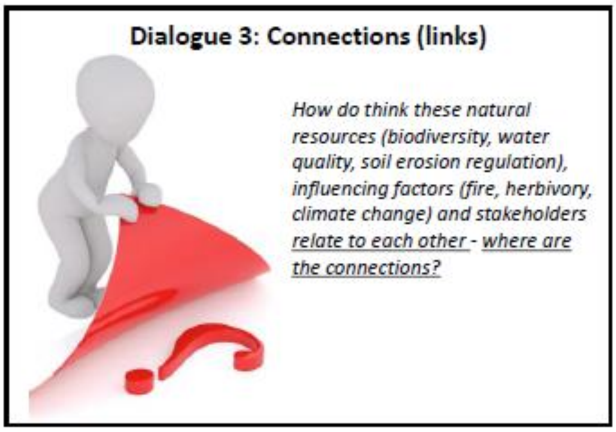


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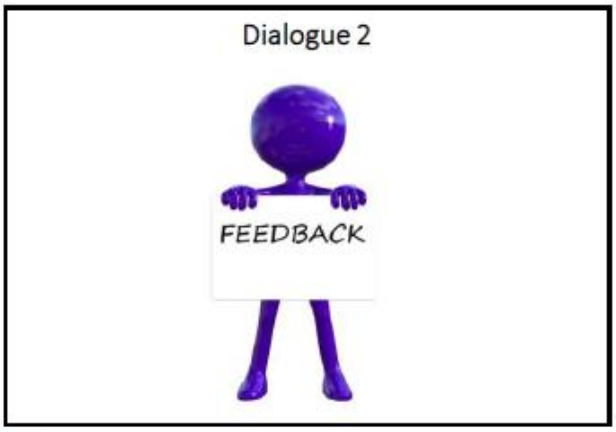


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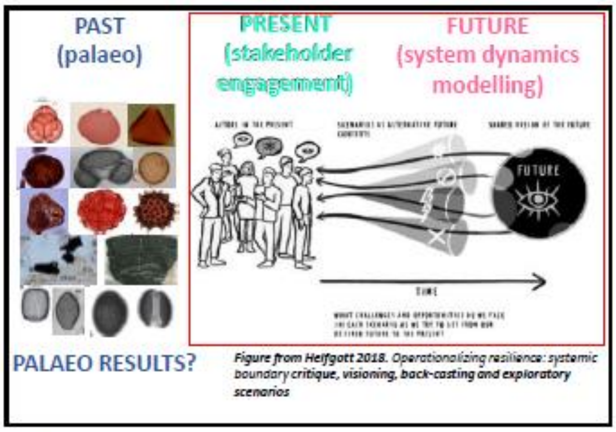


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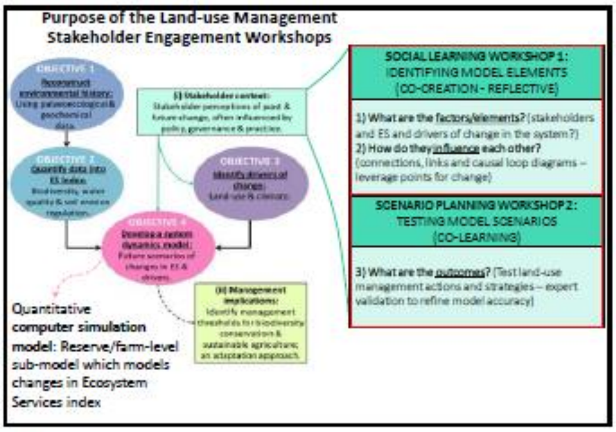


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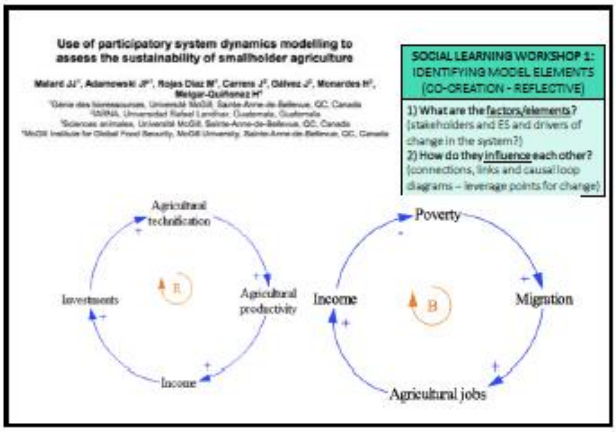


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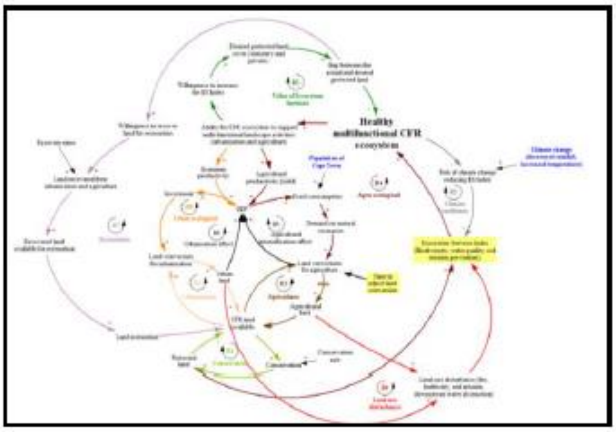


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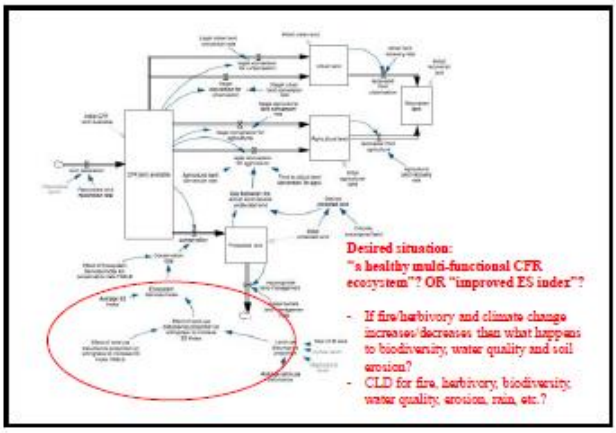


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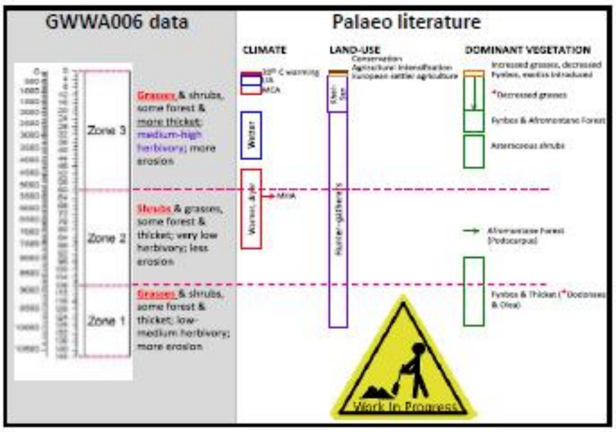


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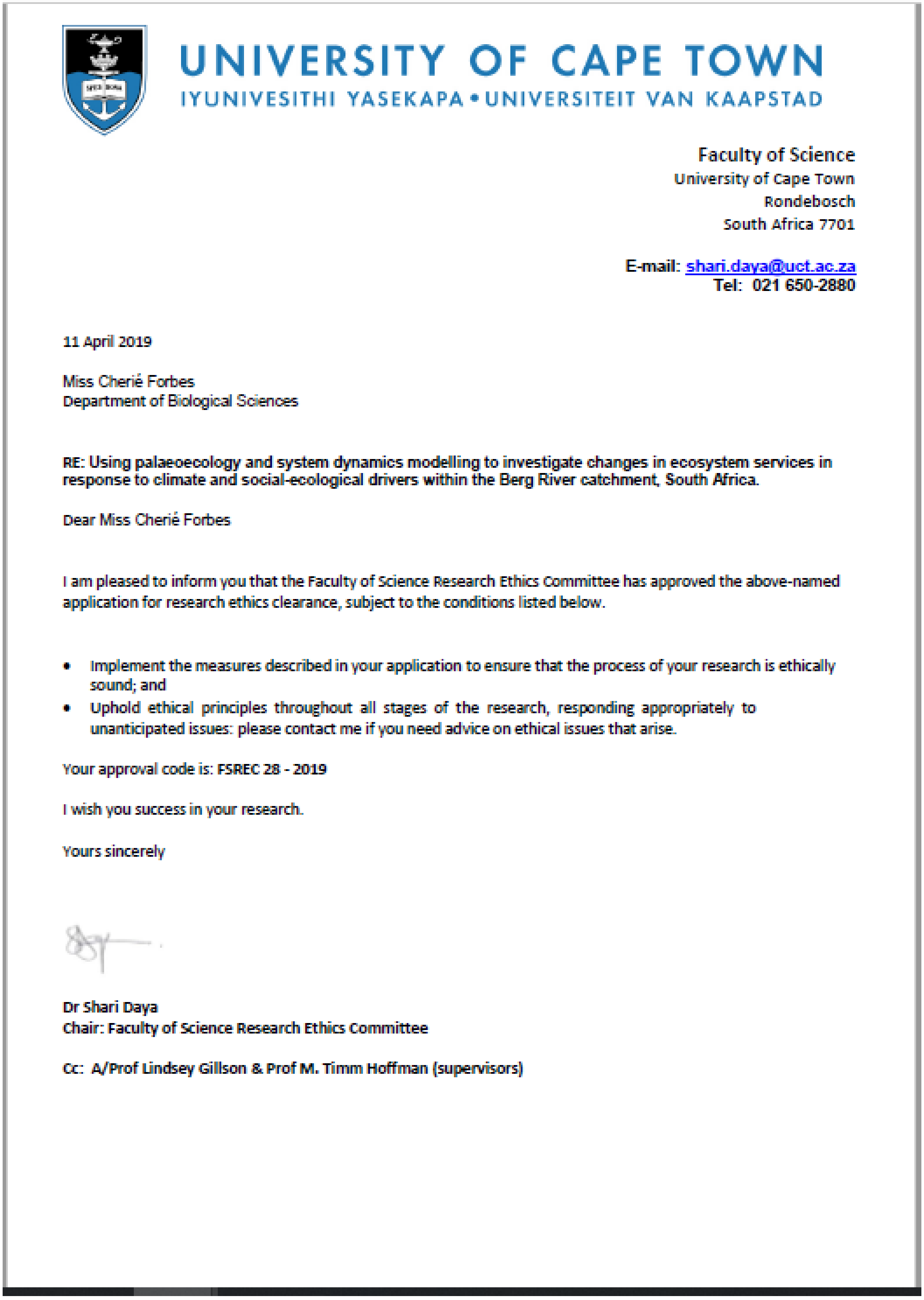
### 10.4.4 Research Ethics Clearance



UNIVERSITY OF CAPE TOWN
IYUNIVESITHI YASEKAPA • UNIVERSITEIT VAN KAAPSTAD

Faculty of Science
University of Cape Town
Rondebosch
South Africa 7701

E-mail: shari.daya@uct.ac.za
Tel: 021 650-2880

11 April 2019

Miss Cherié Forbes
Department of Biological Sciences

**RE: Using palaeoecology and system dynamics modelling to investigate changes in ecosystem services in response to climate and social-ecological drivers within the Berg River catchment, South Africa.**

Dear Miss Cherié Forbes

I am pleased to inform you that the Faculty of Science Research Ethics Committee has approved the above-named application for research ethics clearance, subject to the conditions listed below.

- Implement the measures described in your application to ensure that the process of your research is ethically sound; and
- Uphold ethical principles throughout all stages of the research, responding appropriately to unanticipated issues: please contact me if you need advice on ethical issues that arise.

Your approval code is: **FSREC 28 - 2019**

I wish you success in your research.

Yours sincerely

**Dr Shari Daya**
**Chair: Faculty of Science Research Ethics Committee**

**Cc: A/Prof Lindsey Gillson & Prof M. Timm Hoffman (supervisors)**

*Figure 10.3: Ethics clearance certificate from the Faculty of Science Research Ethics Committee, University of Cape Town (UCT).*

*Consent form for stakeholder engagement– page 1/2*

**DEPARTMENT OF BIOLOGICAL SCIENCES**

UNIVERSITY OF CAPE TOWN
PRIVATE BAG X3
RONDEBOSCH 7701
SOUTH AFRICA

RESEARCHER: Cherie Dirk (Forbes)
TELEPHONE: 0760393659
E-MAIL: cherie.j.forbes@gmail.com
URL: http://www.pcu.uct.ac.za/

**Informed Voluntary Consent to Participate in Research Study**

**Project Title:** *Using palaeoecology and system dynamics modelling to investigate changes in ecosystem services in response to climate and social-ecological drivers within the Lower Berg River catchment, South Africa.*

**Invitation to participate, and benefits:** *You are invited to participate in a research study conducted with commercial farmers, biodiversity conservationists and other land-use governance institutions. The study aim is to use palaeoecology and system dynamics modelling to investigate the long-term influence of climate and land-use change on three key ecosystem services (biodiversity, water quality and soil erosion) dynamics at an upland conservation (Groot Winterhoek Wilderness Area (GWWA)); lowland conservation site (Elandsberg Private Nature Reserve) and a lowland agricultural site (Rhenostervlei farm (RV3)) in the Cape Floristic Region (CFR). Changes in ecosystem services in response to local climate variability and land-use change (fire, herbivory, cultivation and conservation) will be reconstructed over centennial-millennial timescales using palaeoenvironmental-proxies. Semi-structured interviews and workshops with relevant stakeholder groups will be used to investigate perceived variability and resilience of the social-ecological systems and explore implications for land-use management. I believe that your experience would be a valuable source of information, and hope that by participating you may gain useful knowledge.*

**Procedures:** *During this study, you will be asked to participate in two stakeholder workshops as well as a semi-structured interview (optional) relating to how you perceive the relationships between ecosystem services (biodiversity, water quality and soil erosion regulation) and drivers (fire, herbivory, climate), and how these factors change over time (past and future) within the Lower Berg River Catchment of the CFR.*

**Recording:** *As part of the workshop and interview process I would like to make an audio recording of our discussions for verification and analysis purposes only. If you object to this, please indicate this below.*

**Risks:** *There are no potentially harmful risks related to your participation in this study.*

**Disclaimer/Withdrawal:** *Your participation is completely voluntary; you may refuse to participate, and you may withdraw at any time without having to state a reason and without any prejudice or penalty against you. Should you choose to withdraw, the researcher commits not to use any of the information you have provided without your signed consent. Note that the researcher may also withdraw you from the study at any time.*

**Confidentiality:** *All information collected in this study will be kept private in that you will not be identified by name but rather by stakeholder group - i.e. commercial farmer, conservation practitioner, member of land-use governance institution. Confidentiality and anonymity will be maintained as pseudonyms will be used.*

***Figure 10.4: Informed voluntary consent form for research participants complete in order to participate during the stakeholder engagement component of the present study. Ethics Clearance was provided by the University of Cape Town's Faculty of Science Research Ethics Committee. Approval code: FSREC 28. - 2019***

*Consent form for stakeholder engagement– page 2/2*

**What signing this form means:**

*By signing this consent form, you agree to participate in this research study. The aim, procedures to be used, as well as the potential risks and benefits of your participation have been explained verbally to you in detail, using this form. Refusal to participate in or withdrawal from this study at any time will have no effect on you in any way. You are free to contact me, to ask questions or request further information, at any time during this research.*

I agree to participate in this research (tick one box) ☐ Yes ☐ No _________ (Initials)

I agree to be audio-recorded and I agree to the use of properly anonymized audio recordings in publications for research purposes ☐ Yes ☐ No _________ (Initials)

| ____________________ | ____________________ | ________ |
|---|---|---|
| Name of Participant | Signature of Participant | Date |
| ____________________ | ____________________ | ________ |
| Name of Researcher | Signature of Researcher | Date |

## 10.5 Detailed results for GWWA case study

### 10.5.1 Groot Winterhoek Wilderness Area vegetation survey

(a)


*Figure 10.5: Photograph grid of vegetation units which were chosen according to the most dominant vegetation type present on the (a) western slope and (b) eastern slope of where the GWWA006 core was retrieved at the Groot Winterhoek Wilderness Area.*

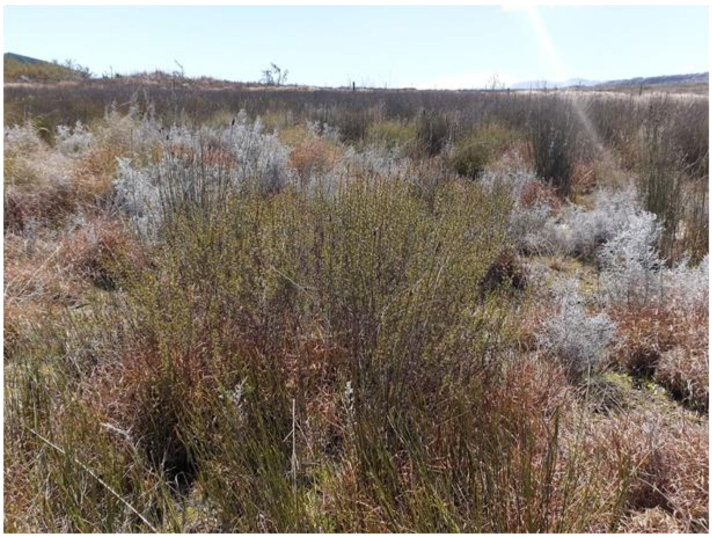
*(Photogrid continued (b) of vegetation unit)*

(b)


.

1. *Typha* (Typhaceae)
2. *Anthochortus crinalis* (Restionaceae)
3. Cyperaceae and Asteraceae

4. *Restio* spp and *Marchantia polymorpha*
5. *Restio* spp



6. *Stoebe* spp, *Erica* spp, *Cliffortia* (other)

***Figure 10.6: Photograph grid of six sections of dominant plant communities along the wetland transect moving from the site where the GWWA006 core was retrieved (1, 2 and 3) in a westward direction.***

***Table 10.3: Modern vegetation survey showing the relative contribution of each taxon within three vegetation units at GWWA006 coring site, Groot Winterhoek Wilderness Area case study site. Estimated percentage cover determined using van der Maarel's (2007) protocols.***

| Vegetation Unit | Taxa (main genera present) | Family | % |
|---|---|---|---|
| VU1a: Taller Restioid shrubland, western sandy slope | *Metalasia* | Asteraceae | 16 |
| | Poaceae | Poaceae | 16 |
| | *Restio* spp #3 | Restionaceae | 10 |
| | *Cliffortia ruschifolia* | Rosaceae | 8 |
| | *Stoebe* spp | Asteraceae | 6 |
| | *Restio* spp #5 | Restionaceae | 5 |
| | *Restio* spp #1 | Restionaceae | 4 |
| | *Restio* spp #2 | Restionaceae | 4 |
| | *Restio* spp #4 | Restionaceae | 4 |
| | *Serruria* spp | Proteaceae | 3 |
| | *Erica* #1 | Ericaceae | 2 |
| | *Leucadendron salignum* | Proteaceae | 2 |
| | *Erica* #2 | Ericaceae | 2 |
| | *Erica* #3 | Ericaceae | 2 |
| | *Erica* #4 | Ericaceae | 2 |
| | *Psammotropha anguina* | Molluginaceae | 2 |
| | *Heeria argentea* | Anacardiaceae | <5% |
| | *Maytenus oleoides* | Celastraceae | <5% |
| | *Searsia* spp | Anacardiaceae | <5% |
| | **Bare ground** | | **14** |
| VU1b: Shorter Restioid shrubland, eastern sandy slope | *Restio* spp #6 | Restionaceae | 22 |
| | *Stoebe* | Asteraceae | 11 |
| | *Restio* spp #1 | Restionaceae | 9 |
| | *Restio* spp #2 | Restionaceae | 8 |
| | *Restio* spp #3 | Restionaceae | 7 |
| | *Asteraceae euryops* | Asteraceae | 5 |
| | *Cliffortia ruschifolia* | Rosaceae | 5 |
| | *Cliffortia (other)* | Rosaceae | 5 |
| | *Restio* spp #7 | Restionaceae | 2 |
| | *Rumex bucephalophorus* | Polygonaceae | 2 |
| | *Anthochortus crinalis* | Restionaceae | 2 |
| | *Metalasia* spp | Asteraceae | 0 |
| | *Podalyria* | Fabaceae | <5% |
| | *Myrsine africana* | Primulaceae | <5% |
| | *Dyospyros* | Ebenaceae | <5% |
| | *Rhus/ Searsia spp* | Anacardiaceae | <5% |
| | *Dodonaea viscosa* | Sapindaceae | <5% |
| | *Maytenus oleoides* | Celastraceae | <5% |
| | *Pelargonium* | Geraniaceae | <5% |
| | *Lachenalia* | Asparagaceae | <5% |

| | | | |
|---|---|---|---|
| | Bruniaceae | Bruniaceae | <5% |
| | **Bare ground** | | **45** |
| GWWA006 Wetland transect | *Typha* | Typhaceae | |
| | *Anthochortus crinalis* | Restionaceae | |
| | *Cyperaceae* | Cyperaceae | |
| | *Restio* spp | Restionaceae | |
| | *Marchantia polymorpha* | Marchantiaceae | |
| | *Stoebe* spp | Asteraceae | |
| | *Erica* spp | Ericaceae | |
| | *Cliffortia (other)* | Rosaceae | |
| | *Drosera* spp | Droseraceae | |
| | Poaceae (several spp) | Poaceae | |

### 10.5.2 Appendix: Radiocarbon dating

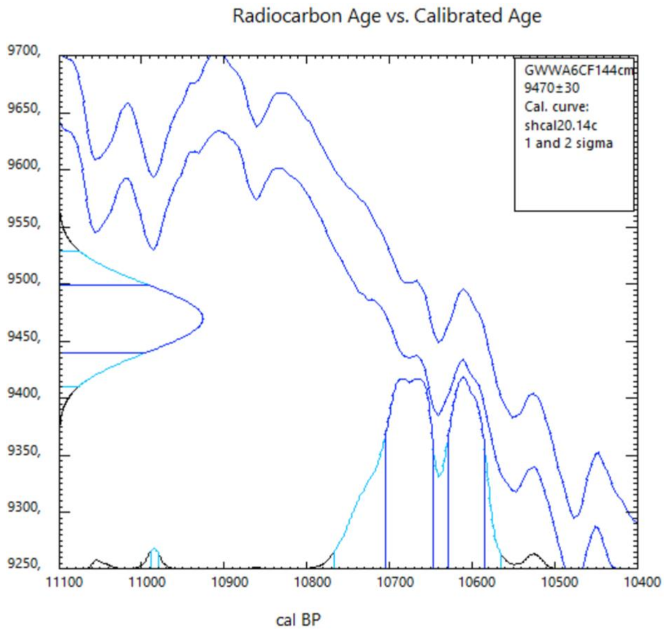



***Figure 10.7: Highest posterior density ranges for basal radiocarbon date from GWWA006 core at 144 cm.***

***Table 10.4: Results obtained from radiocarbon dating of eight samples from the GWWA006 core. Modelled calibrated age was calculated according to the Bacon age-depth model.***

| **Sample name /laboratory code** | **Sample depth (cm)** | **Radiocarbon age (yr BP)** | **BCal Calibrated date (cal yr BP)** | **Probability (Standard deviation and %)** | **Modelled calibrated age (cal yr BP)** |
|---|---|---|---|---|---|
| GWWA6CF0cm | 0 | 730±97 | 682-622 | 2 SD (95%) | 1032 |
| GWWA6CF4cm | 4 | 1810±44 | 1748-1694 | 2 SD (95%) | 1589 |
| GWWA6CF20cm | 20 | 2150±38 | 2116-2039 | 2 SD (95%) | 2197 |
| GWWA6CF56cm | 56 | 4180±45 | 4726-4613 | 2 SD (95%) | 4698 |
| GWWA6CF88cm | 88 | 6870±53 | 7714-7606 | 2 SD (95%) | 7659 |
| GWWA6CF124cm | 124 | 9290±64 | 10 509-10 335 | 2 SD (95%) | 10 001 |
| GWWA6CF132cm | 132 | 9210±90 | 10 423-10 240 | 2 SD (95%) | 10 364 |
| GWWA6CF144cm | 144 | 9470±30 | 10 706-10 647 | 2 SD (95%) | 10 764 |

### 10.5.3 Appendix: Pollen zonation broken stick graphs and dendograms

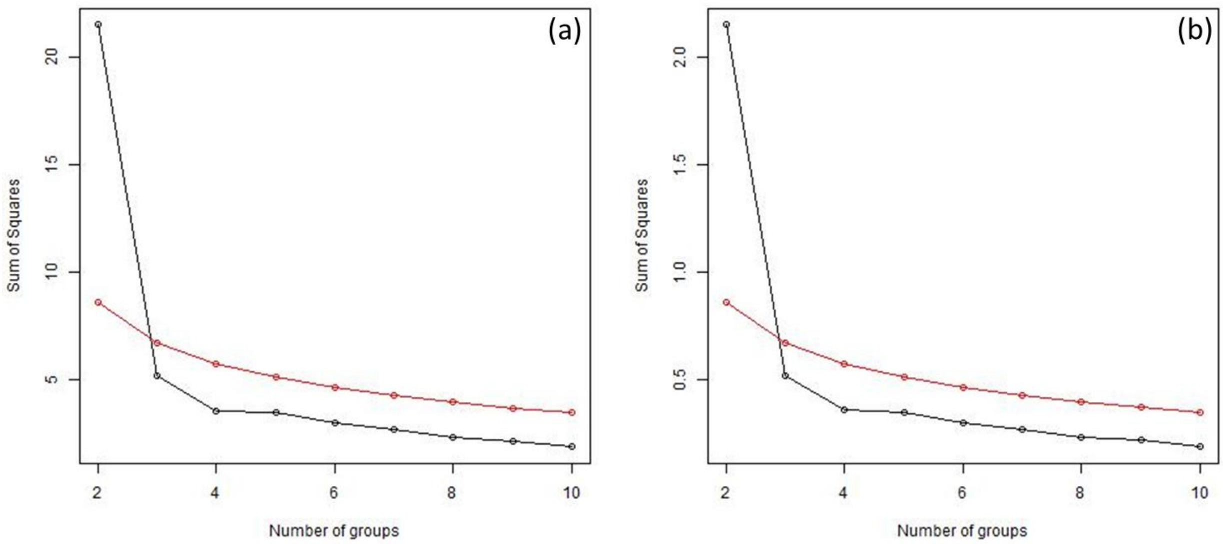



*Figure 10.8: GWWA006 broken stick models to determine the number of statistically significant zones in the cluster analysis of the (a) percentage and (b) concentration pollen stratigraphic data from the GWWA006 core. The black line indicates model simulations based on the empirical data, and the red represents random simulations.*

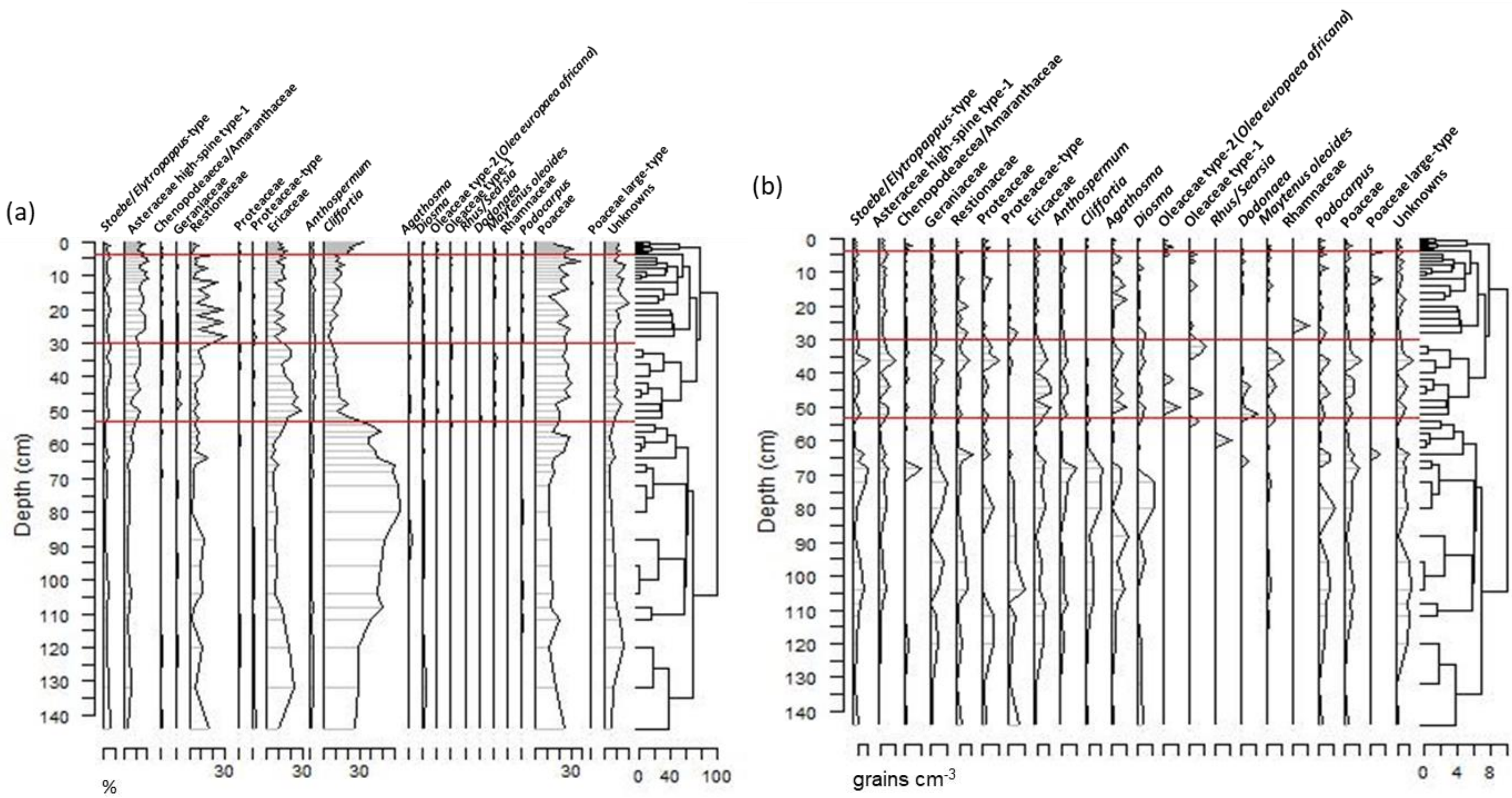



*Figure 10.9: GWWA006 dendrogram produced by a cluster analysis showing distribution of (a) percentage and (b) concentration pollen taxa based on depth of the GWWA006 core from Groot Winterhoek Wilderness Area. The depth factor was classified into sub-sones. Cluster analysis based on square root transformed and Bray-Curtis similarity data.*

### 10.5.4 Appendix: NMDS ordination stress values and plot

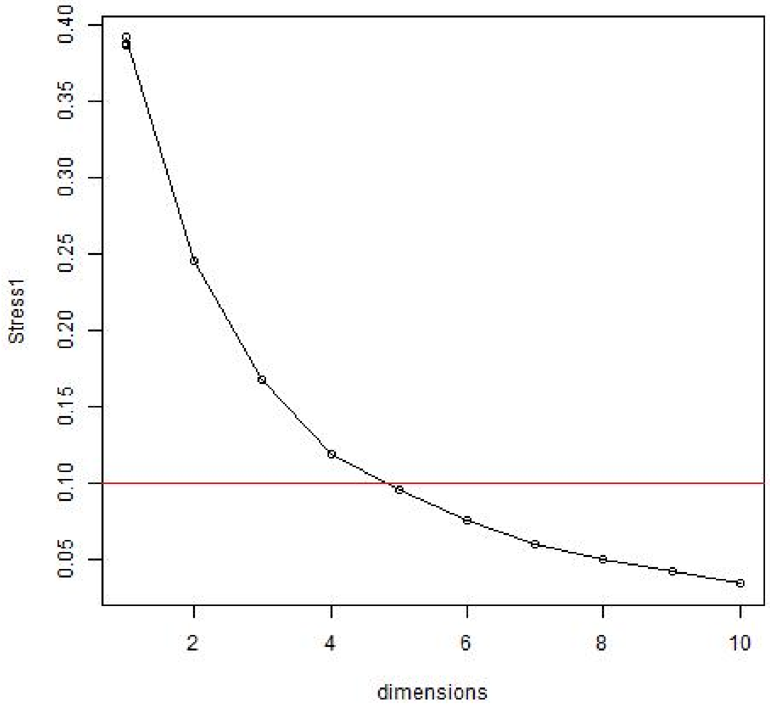



*Figure 10.10: Stress values for the NMDS ordination of the GWWA006 core. The number of dimensions was 5*

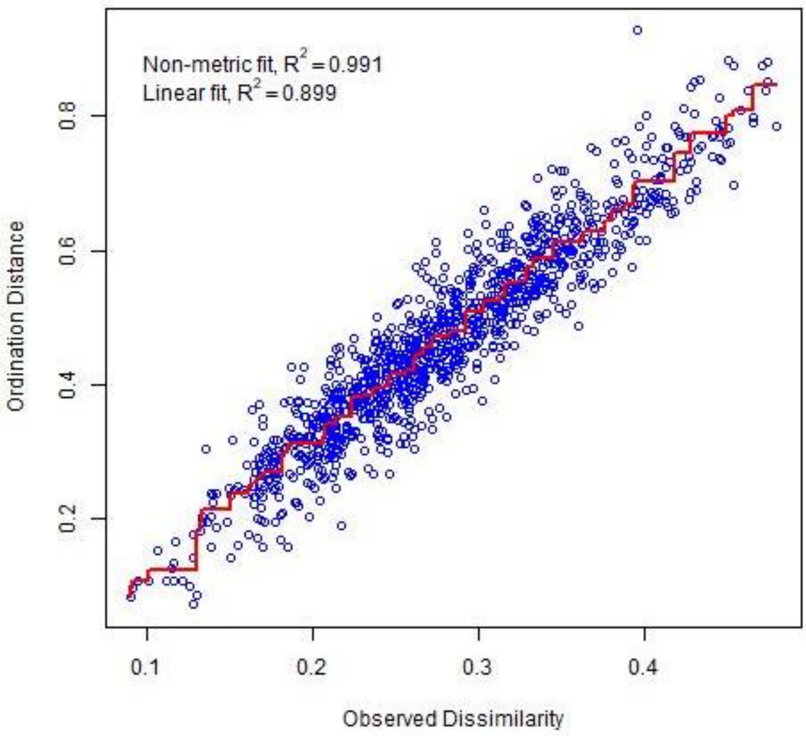



*Figure 10.11: Stress plot for the NMDS ordination of the GWWA006 core. Linear fit $R^2$ 0.89.*

## 10.6 Detailed results for RV3 case study

### 10.6.1 Appendix: Rhenostervlei Farm Vegetation Survey


***Figure 10.12: Photograph grid of vegetation units which were chosen according to the most dominant vegetation type present at the Rhenostervlei Farm study site where the RV3 core was retrieved.***

***Table 10.5: Modern vegetation survey showing the relative contribution of each taxon within five vegetation units at RV3, Rhenostervlei Farm case study site. Estimated percentage cover determined using van der Maarel's (2007) protocols.***

| Vegetation Unit | Taxa (main genera present) | Family | % |
|---|---|---|---|
| VU 1: Riverine fringe (RV3 wetland located) | *Eucalyptus camaldulensis* | Myrtaceae | 79 |
| | *Searsia* | Anacardiaceae | 31 |
| | *Ehrharta longifolia* | Poaceae | 26 |
| | *Lolium perenne* | Poaceae | 26 |
| | *Cynodon dactylon* | Poaceae | 22 |
| | *Olea europaea africana* | Oleaceae | 25 |
| | *Stellaria media* | Caryophyllaceae | 6 |
| | *Kiggelaria africana* | Achariaceae | 6 |
| | *Leonotis leonurus* | Lamiaceae | 6 |
| | *Ranunculus muricatus* | Ranunculaceae | 3 |
| | *Schoenoplectus spp* | Cyperaceae | 3 |
| | *Cotula coronopifolia* | Asteraceae | 3 |
| | *Zantedeschia aethiopica* | Araceae | 3 |
| | *Urtica urens* | Urticaceae | 2 |
| | *Oxalis* spp | Oxalidaceae | 2 |
| | *Calystegia* spp | Convolvulaceae | 2 |
| | *Cyperus* spp | Cyperaceae | 1 |
| | *Sonchus oleraceus* | Asteraceae | 3 |
| | *Hypochaeris radicata* | Asteraceae | 3 |
| | *Oxalis pes-caprae* | Oxalidaceae | 3 |
| | *Diospyros* spp | Ebenaceae | 2 |
| | *Pterygodium orobanchoides* | Orchidaceae | 1 |
| | *Prionium serratum* | Thurniaceae | 1 |
| | *Anagallis monelli* | Primulaceae | 1 |
| | **Bare ground** | | **42** |
| VU 2: Floodplain | *Ehrharta longiflora* | Poaceae | 42 |
| | *Isolepis* spp | Cyperaceae | 35 |
| | *Schoenoplectus* | Cyperaceae | 31 |
| | *Vulpia myurus* | Poaceae | 13 |
| | *Poaceae* spp | Poaceae | 9 |
| | *Diospyros* spp | Ebenaceae | 6 |
| | *Cynodon dactylon* | Poaceae | 6 |
| | *Geranium caffrum* | Geraneaceae | 6 |
| | *Ehrharta calycina* | Poaceae | 3 |
| | *Briza maxima* | Poaceae | 3 |
| | *Oxalis* spp | Oxalidaceae | 3 |
| | *Briza minor* | Poaceae | 3 |
| | **Bare ground** | | **1** |
| VU 3: Renosterveld Slope | *Drosanthemum asperulum* | Aizoaceae | 38 |
| | *Coleonema/Agathosma* | Rutaceae | 15 |
| | *Elytropappus rhinocerotis* | Asteraceae | 8 |
| | *Passerina* spp | Thymelaeaceae | 3 |

| | | | |
|---|---|---|---|
| | *Phylica* spp | Rhamnaceae | 1 |
| | **Bare ground** | | **46** |
| VU 4: Tall Renosterveld Shrubland | *Passerina* spp | Thymelaeaceae | 40 |
| | *Spergula* arvensis | Caryophyllaceae | 27 |
| | *Elytropappus rhinocerotis* | Asteraceae | 15 |
| | *Montinia caryophyllacea* | Montiniaceae | 13 |
| | *Ursinia anthemoides* | Asteraceae | 12 |
| | *Asteraceae* spp | Asteraceae | 5 |
| | *Phagnalon* spp | Asteraceae | 3 |
| | *Drosanthemum asperulum* | Aizoaceae | 3 |
| | *Cliffortia* spp | Rosaceae | 3 |
| | *Eriocephalus africanus* | Asteraceae | 3 |
| | *Restio* spp | Restionaceae | 2 |
| | **Bare ground** | | **19** |
| VU 5: Sandstone Fynbos sand dunes - transect | *Leucadendron* spp | Proteaeceae | |
| | *Leucospermum rodolentum* | Proteaeceae | |
| | *Diospyros* spp | Ebenaceae | |
| | *Passerina* spp | Thymelaeaceae | |
| | *Restio* spp | Restionaceae | |
| | *Olea europaea africana* | Oleaceae | |
| | *Acacia saligna* | Fabaceae | |
| | **Bare ground** | | **N/A** |

### 10.6.2 Appendix: Radiocarbon dating

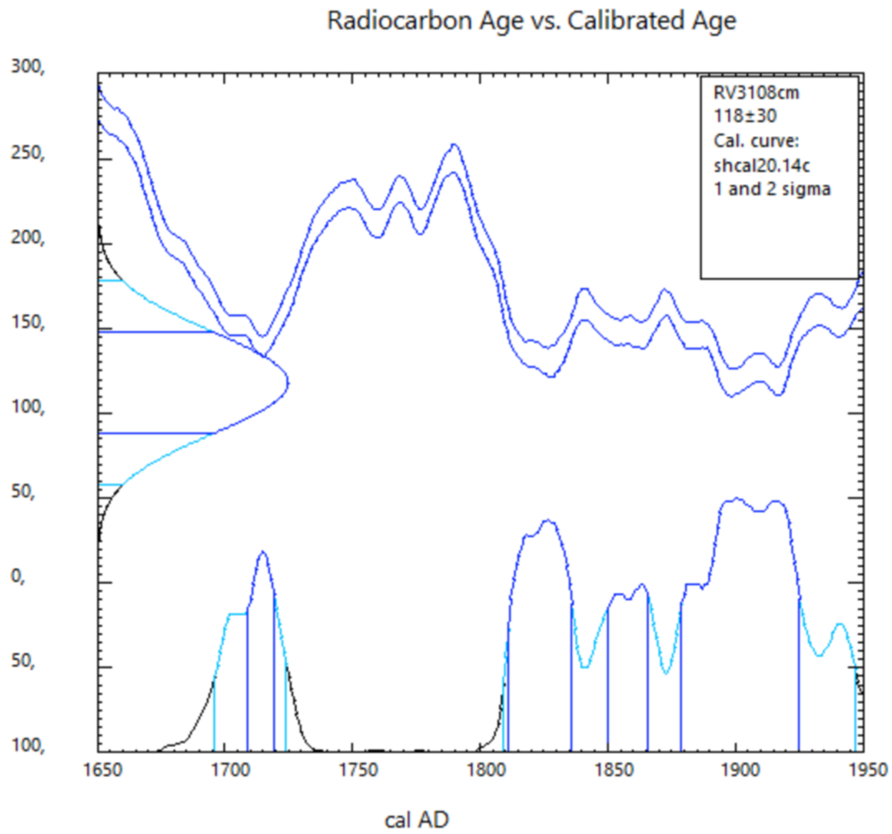



***Figure 10.13: Highest posterior density ranges for one radiocarbon date from RV3 core at 108 cm.***

### 10.6.3 Appendix: Lead-210 dating of RV3 core

***Table 10.6: Results obtained from (a) $^{210}$Pb dating of thirty-three samples and (b) radiocarbon dating of one sample from the RV3 core. Results for sediment sections no. 32 and 33 are not shown since they were older than 150 years.***

| a) | Sediment section no. | Sample depth (cm) | Age at Bottom of Extrapolated Section in yr BP (constant rate of supply CRS model estimate) | CRS Sediment Accumulation Rate ($g/cm^2/yr$) | $^{210}$Pb Total Activity (DPM/g) | $^{210}$Pb Unsupported Activity (DPM/g) | Error $^{210}$Pb +/- 1 S. D. (DPM/g) | Calendar years (AD) |
|---|---|---|---|---|---|---|---|---|
| | 1 | 0-2 | 1.60 | 0.271 | 13.772 | 12.652 | 0.47 | 2009 |
| | 2 | 2-4 | 3.29 | 0.266 | 13.562 | 12.442 | 0.47 | 2008 |
| | 3 | 4-6 | 5.11 | 0.260 | 13.530 | 12.410 | 0.47 | 2006 |
| | 4 | 6-8 | 6.95 | 0.258 | 12.608 | 11.488 | 0.45 | 2004 |
| | 5 | 8-10 | 8.93 | 0.254 | 12.507 | 11.387 | 0.45 | 2002 |
| | 6 | 10-12 | 10.90 | 0.253 | 11.593 | 10.473 | 0.43 | 2000 |
| | 7 | 12-14 | 12.88 | 0.252 | 10.886 | 9.766 | 0.42 | 1998 |
| | 8 | 14-16 | 14.93 | 0.251 | 10.383 | 9.263 | 0.41 | 1996 |
| | 9 | 16-18 | 17.07 | 0.250 | 10.005 | 8.885 | 0.4 | 1994 |
| | 10 | 18-20 | 19.26 | 0.249 | 9.421 | 8.301 | 0.39 | 1992 |
| | 11 | 20-22 | 21.57 | 0.247 | 9.170 | 8.050 | 0.38 | 1989 |
| | 12 | 22-24 | 23.93 | 0.245 | 8.628 | 7.508 | 0.37 | 1987 |
| | 13 | 24-26 | 26.34 | 0.244 | 8.048 | 6.928 | 0.36 | 1985 |
| | 14 | 26-28 | 28.84 | 0.243 | 7.694 | 6.574 | 0.35 | 1982 |
| | 15 | 28-30 | 31.44 | 0.241 | 7.325 | 6.205 | 0.34 | 1980 |
| | 16 | 30-33 | 35.34 | 0.239 | 6.685 | 5.565 | 0.33 | 1976 |
| | 17 | 33-36 | 39.39 | 0.237 | 6.164 | 5.044 | 0.32 | 1972 |
| | 18 | 36-39 | 43.58 | 0.235 | 5.634 | 4.514 | 0.3 | 1967 |
| | 19 | 39-42 | 48.00 | 0.232 | 5.238 | 4.118 | 0.29 | 1963 |
| | 20 | 42-45 | 52.53 | 0.230 | 4.745 | 3.625 | 0.28 | 1958 |
| | 21 | 45-48 | 57.26 | 0.227 | 4.385 | 3.265 | 0.27 | 1954 |
| | 22 | 48-51 | 62.31 | 0.223 | 4.086 | 2.966 | 0.26 | 1949 |
| | 23 | 51-54 | 67.64 | 0.220 | 3.757 | 2.637 | 0.25 | 1943 |
| | 24 | 54-57 | 73.16 | 0.216 | 3.384 | 2.264 | 0.24 | 1938 |
| | 25 | 57-60 | 78.89 | 0.213 | 3.076 | 1.956 | 0.22 | 1932 |
| | 26 | 60-65 | 88.49 | 0.208 | 2.636 | 1.516 | 0.21 | 1923 |
| | 27 | 65-70 | 100.52 | 0.200 | 2.453 | 1.333 | 0.2 | 1910 |
| | 28 | 70-75 | 115.77 | 0.189 | 2.216 | 1.096 | 0.19 | 1895 |
| | 29 | 75-80 | 136.20 | 0.173 | 1.959 | 0.839 | 0.18 | 1875 |
| | 30 | 80-85 | 157.39 | 0.161 | 1.566 | 0.446 | 0.16 | 1854 |
| | 31 | 85-90 | 184.92 | 0.147 | 1.391 | 0.271 | 0.15 | 1826 |
| | 32 | 90-95 | - | - | - | - | - | - |
| | 33 | 95-100 | - | - | - | - | - | - |
| | | | | **mean 0.2301** | | | | |

| b) | Sample depth (cm) | Radiocarbon date (yr BP) | Calibrated date (AD) | Probability (Standard deviation and %) | Modelled calibrated age (AD) |
|---|---|---|---|---|---|
| | 108 | 118 ± 30 | 1879-1925 | 2 SD (95%) | **1795** |

The Slope Regression model for Lead-210 dating of RV3 included sediment sections 1-32 (0-95 cm) and assumed a constant sediment accumulation and $^{210}Pb$ input. An associated $R^2$ table was used to determine the sediment accumulation rate where the $R^2$ values provide a suitable fit as a function of the background level of $^{210}Pb$ which was subtracted (Table 10.7). $R^2$ values showed equally good fits for a large range of background $^{210}Pb$ values in the RV3 core. It was assumed that the lowest observed activity (1.12 DPM/g; where DPM is the abbreviation for disintegration per minute) of the core at sediment section 33 (95-100 cm) was the true background $^{210}Pb$ level. Hence the closest corresponding sediment accumulation rate in the $R^2$ table ($R^2$ = 0.9778) was about 0.2203 g/cm$^2$/yr.

***Table 10.7: Table with $R^2$ values and corresponding sediment accumulation rates. $R^2$ values as a function of background $^{210}Pb$ level subtracted. Note that the $R^2$ value and corresponding accumulation rate associated with the RV3 core is bolded and marked with an asterisk (*).***

| ***Background $^{210}Pb$ level* (DPM/g)** | **$R^2$** | **Sediment Accumulation Rate (g/cm$^2$/yr)** | **Slope 'm'** | **Y intercept 'b'** |
|---|---|---|---|---|
| 0.0000 | 0.9961 | 0.3458 | -11.1264 | 29.4645 |
| 0.0555 | 0.9965 | 0.3410 | -10.9709 | 29.0528 |
| 0.1109 | 0.9969 | 0.3361 | -10.8134 | 28.6384 |
| 0.1664 | 0.9973 | 0.3311 | -10.6535 | 28.2210 |
| 0.2218 | 0.9977 | 0.3261 | -10.4911 | 27.8003 |
| 0.2773 | 0.9980 | 0.3209 | -10.3260 | 27.3759 |
| 0.3327 | 0.9983 | 0.3157 | -10.1578 | 26.9474 |
| 0.3882 | 0.9985 | 0.3104 | -9.9864 | 26.5143 |
| 0.4436 | 0.9986 | 0.3049 | -9.8112 | 26.0762 |
| 0.4991 | 0.9986 | 0.2994 | -9.6320 | 25.6322 |
| 0.5545 | 0.9985 | 0.2937 | -9.4482 | 25.1818 |
| 0.6100 | 0.9983 | 0.2878 | -9.2592 | 24.7238 |
| 0.6655 | 0.9979 | 0.2817 | -9.0643 | 24.2572 |
| 0.7210 | 0.9973 | 0.2755 | -8.8625 | 23.7806 |
| 0.7765 | 0.9965 | 0.2689 | -8.6527 | 23.2921 |
| 0.8320 | 0.9953 | 0.2621 | -8.4334 | 22.7895 |
| 0.8875 | 0.9936 | 0.2549 | -8.2027 | 22.2698 |
| 0.9430 | 0.9913 | 0.2473 | -7.9577 | 21.7286 |
| 0.9985 | 0.9882 | 0.2392 | -7.6946 | 21.1602 |
| 1.0540 | 0.9840 | 0.2302 | -7.4075 | 20.5555 |
| **1.1095*** | **0.9778*** | **0.2203*** | **-7.0867*** | **19.9000*** |

There is an approximately exponential decrease in the $^{210}Pb$ activity as a function of depth in the RV3 core (Figure 10.14). The surface level activity of $^{210}Pb$ (i.e. 13.77 DPM/g of sediment section no. 1; 0-2 cm) was approximately 12 times greater than the observed background activity (i.e. 1.12 DPM/g; the lowest observed activity of the RV3 core).

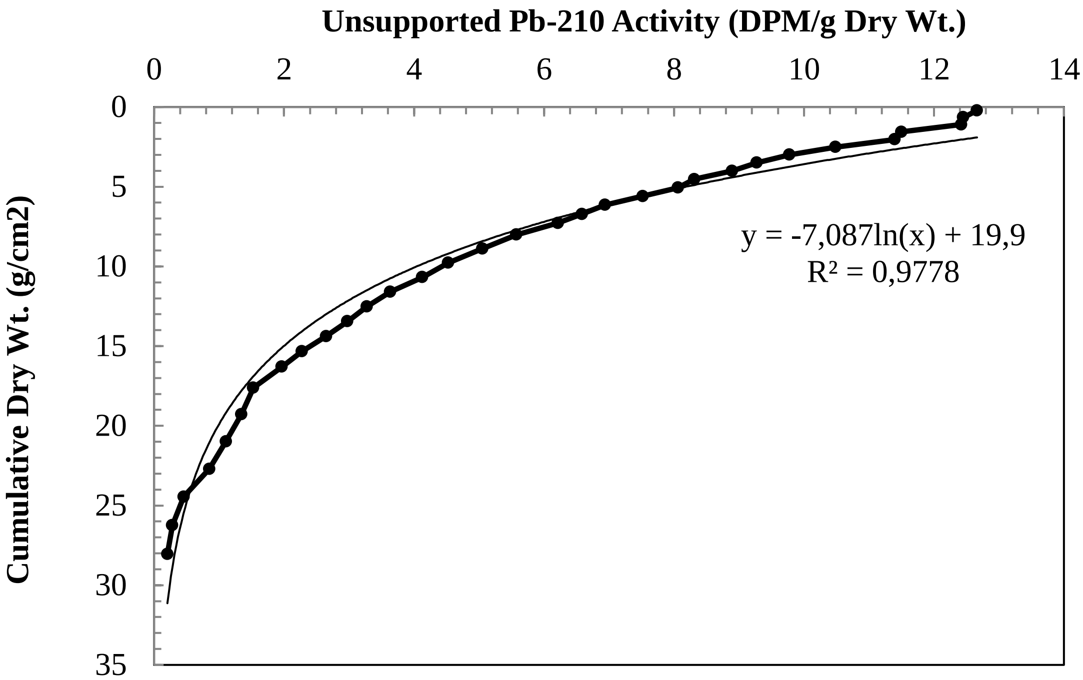



*Figure 10.14: Regression of Unsupported $^{210}$Pb Activity and Accumulated Sediment using $^{210}$Pb background of 1.1095 DPM/g.*

### 10.6.4 Appendix: Pollen and Diatom Zonation broken stick graphs and dendograms

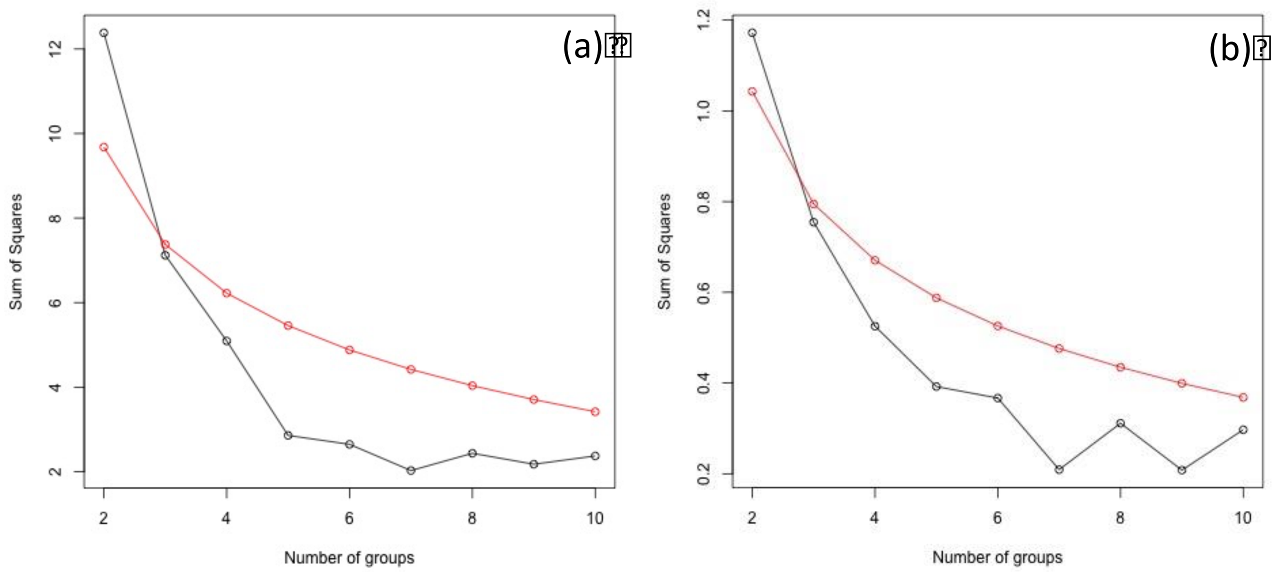



*Figure 10.15: Broken stick models to determine the number of statistically significant zones in the cluster analysis of the (a) percentage and (b) concentration pollen stratigraphic data from the RV3 core. The black line indicates model simulations based on the empirical data, and the red represents random simulations.*

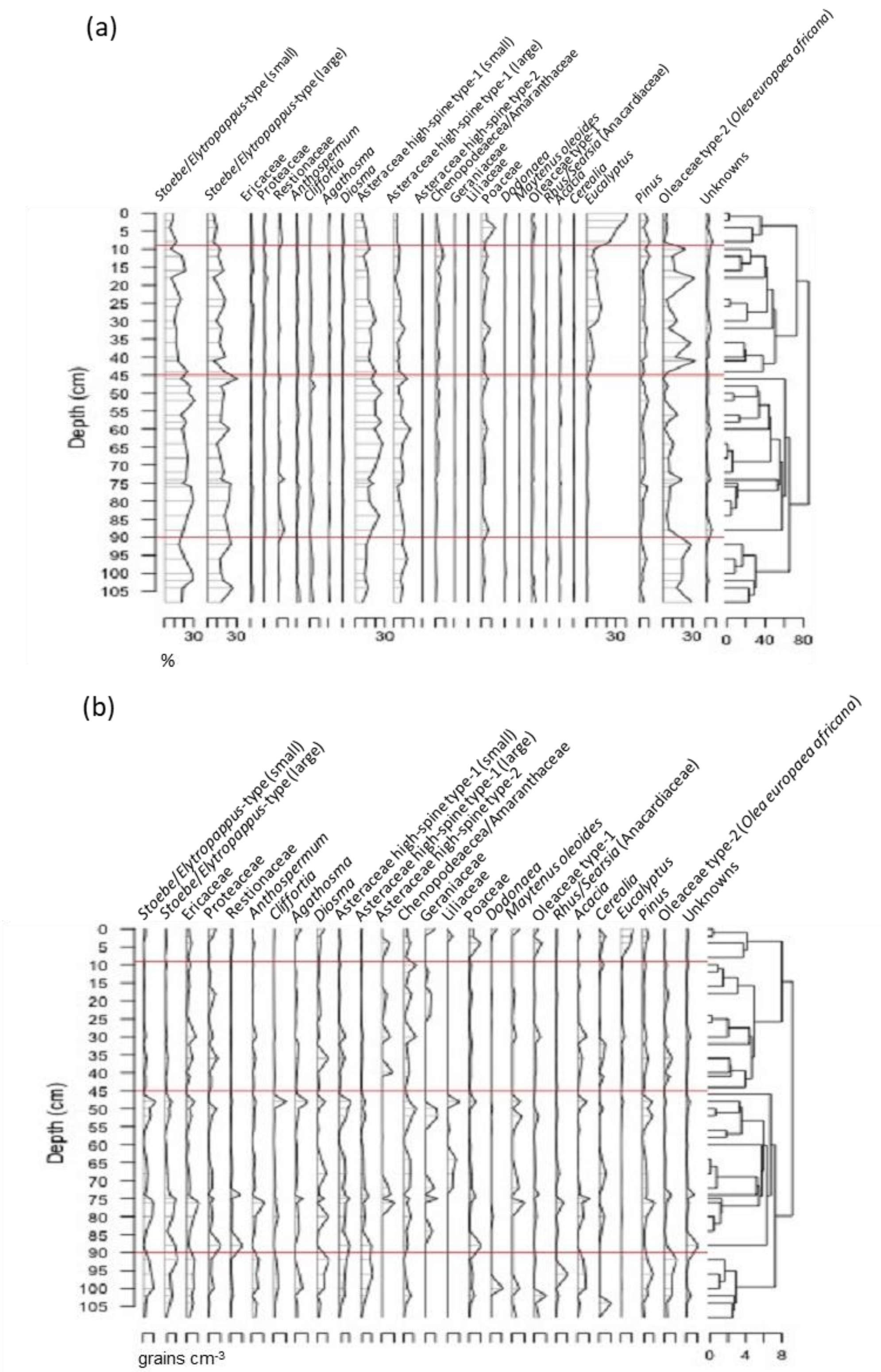



***Figure 10.16: Dendrogram produced by a cluster analysis showing distribution of (a) percentage and (b) concentration pollen taxa based on depth of the RV3 core from Rhenostervlei Farm. The depth factor was classified into sub-sones. Cluster analysis based on square root transformed and Bray-Curtis similarity data.***

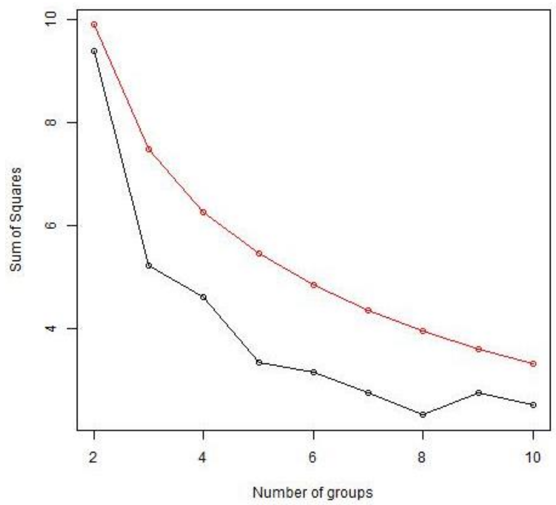



*Figure 10.17: Broken stick model to determine the number of significant zones in the cluster analysis of the diatom percentage stratigraphic data from the RV3 core. The black line indicates model simulations based on the empirical data, and the red represents random simulations.*

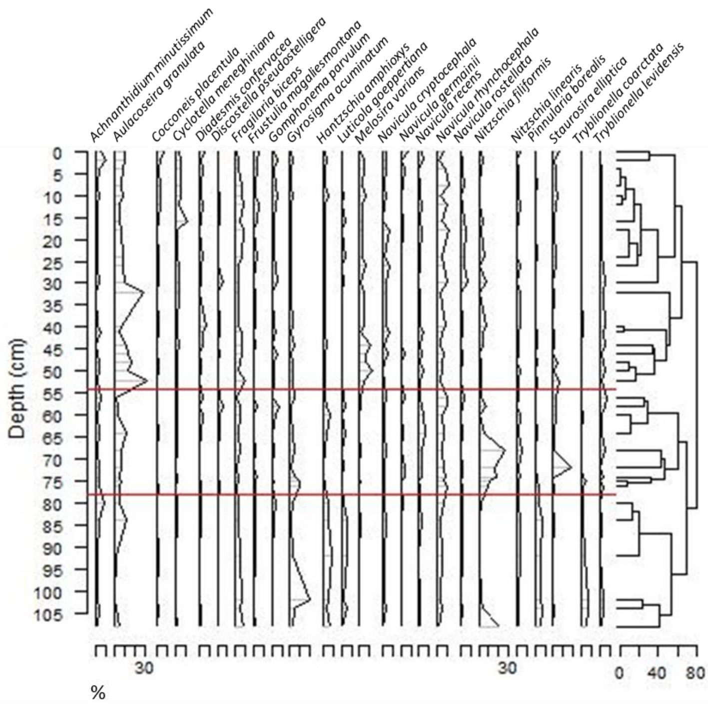



*Figure 10.18: Dendrogram produced by a cluster analysis showing distribution of percentage diatom taxa based on depth for the RV3 core from Rhenostervlei Farm. Cluster analysis based on square root transformed and Bray-Curtis similarity data. The depth factor was classified into three different sub- zones: A (108-78 cm), B (78-54 cm) and C (54-0 cm) which are separated by a dashed line to show that the zones are not statistically significant.*

### 10.6.5 Appendix: Unabridged percentage diatom diagrams for RV3 core

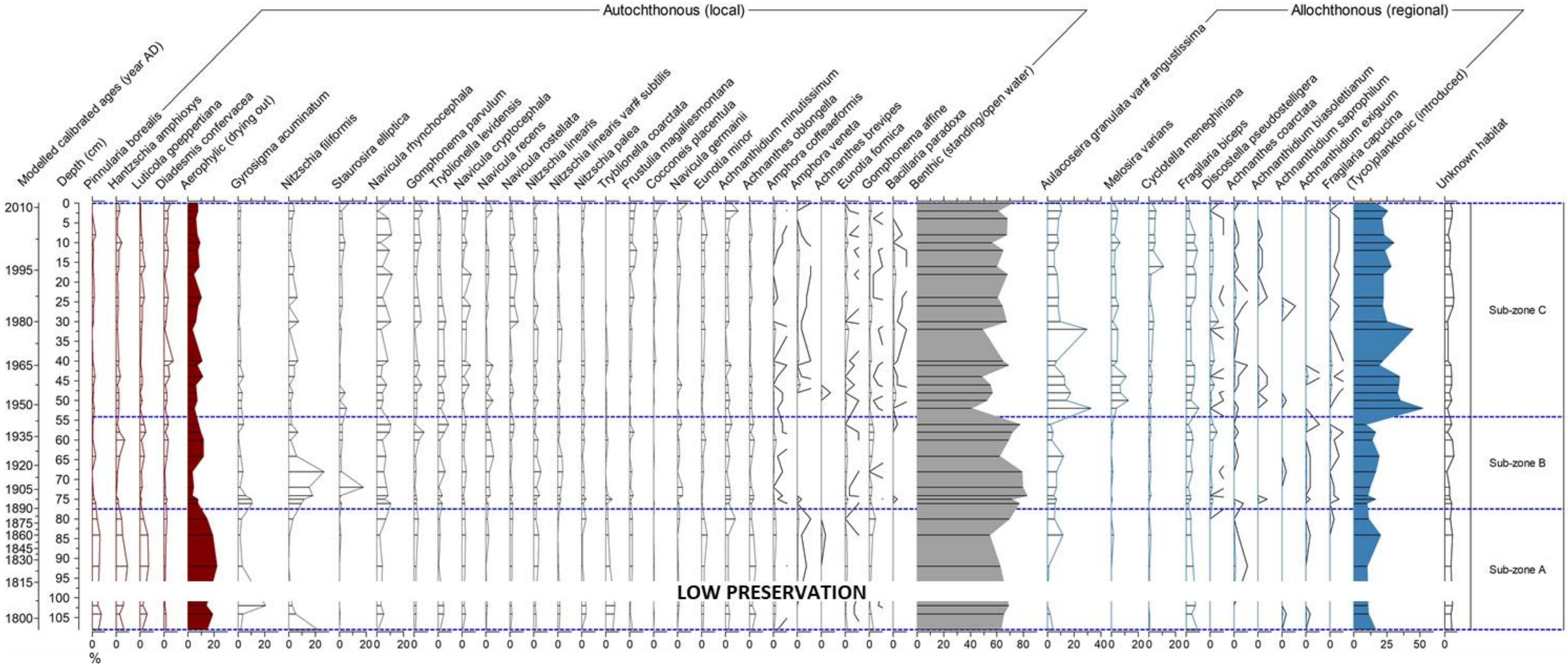



***Figure 10.19: Percentage diatom diagram with selected diatom taxa that depict habitat preference from the RV3 core at Rhenostervlei Farm, Middle Berg River Catchment, South Africa. x10 exaggeration is shown since percentage values are often low for most taxa. Summary graphs with diatom taxa grouped into ecological groupings of habitat preferences (drying out, standing/open water and introduced from the river). Zonation is calculated by cluster analysis based on 70% minimum resemblance levels –dashed lines show sub-zones (A, B, C) that were not statistically significant to the far right of the diagram. Modelled calibrated ages which were determined using an age-depth model (Bacon)are shown on the left. Low diatom preservation is shown between 96-100 cm in Sub-zone A***

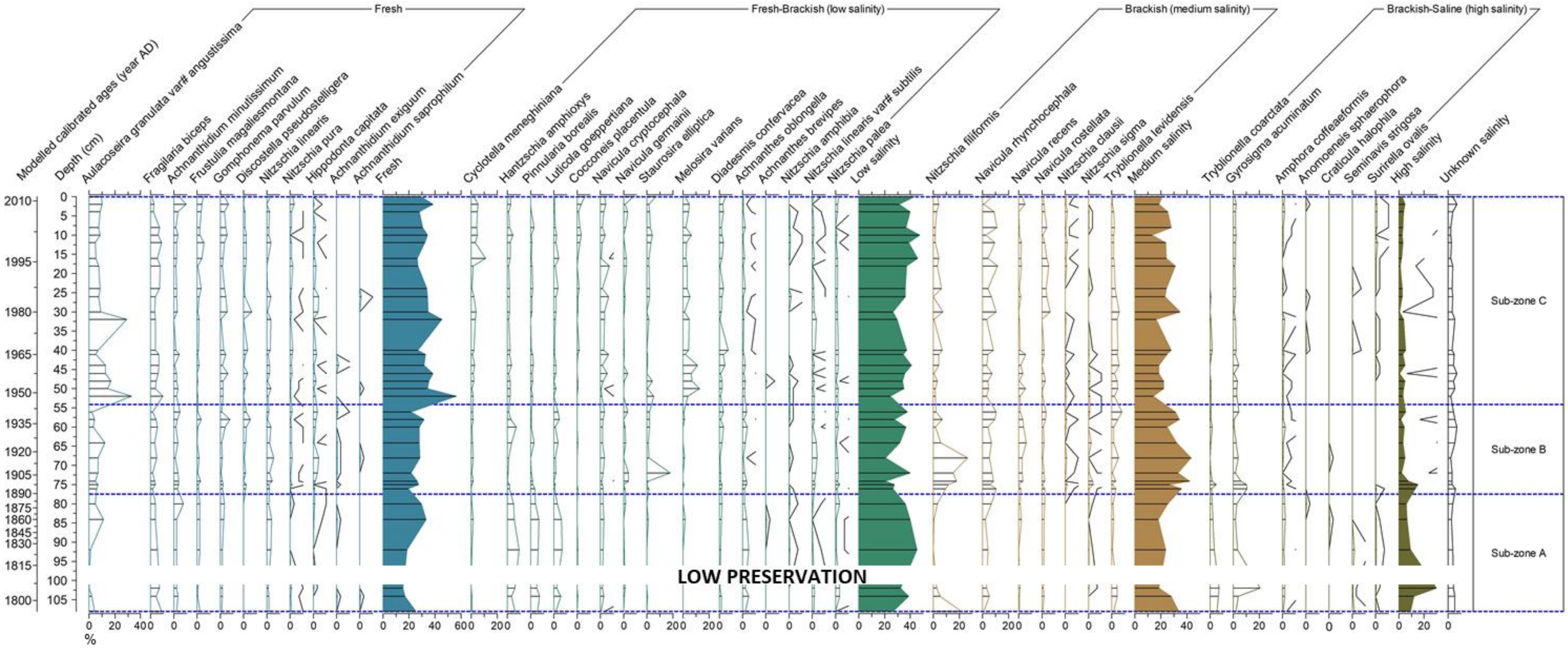



***Figure 10.20: Percentage diatom diagram with selected diatom taxa that depict salinity preference from the RV3 core at Rhenostervlei Farm, Middle Berg River Catchment, South Africa.. x10 exaggeration is shown since percentage values are often low for most taxa. Diatom taxa are grouped into salinity preferences (fresh, low, medium and high salinity). Although not statistically significant, sub-zones (A, B, C) are indicated by the dashed lines. Modelled calibrated ages are shown on the far left of the diagram. Low diatom preservation is shown between 96-100 cm in Sub-zone A.***

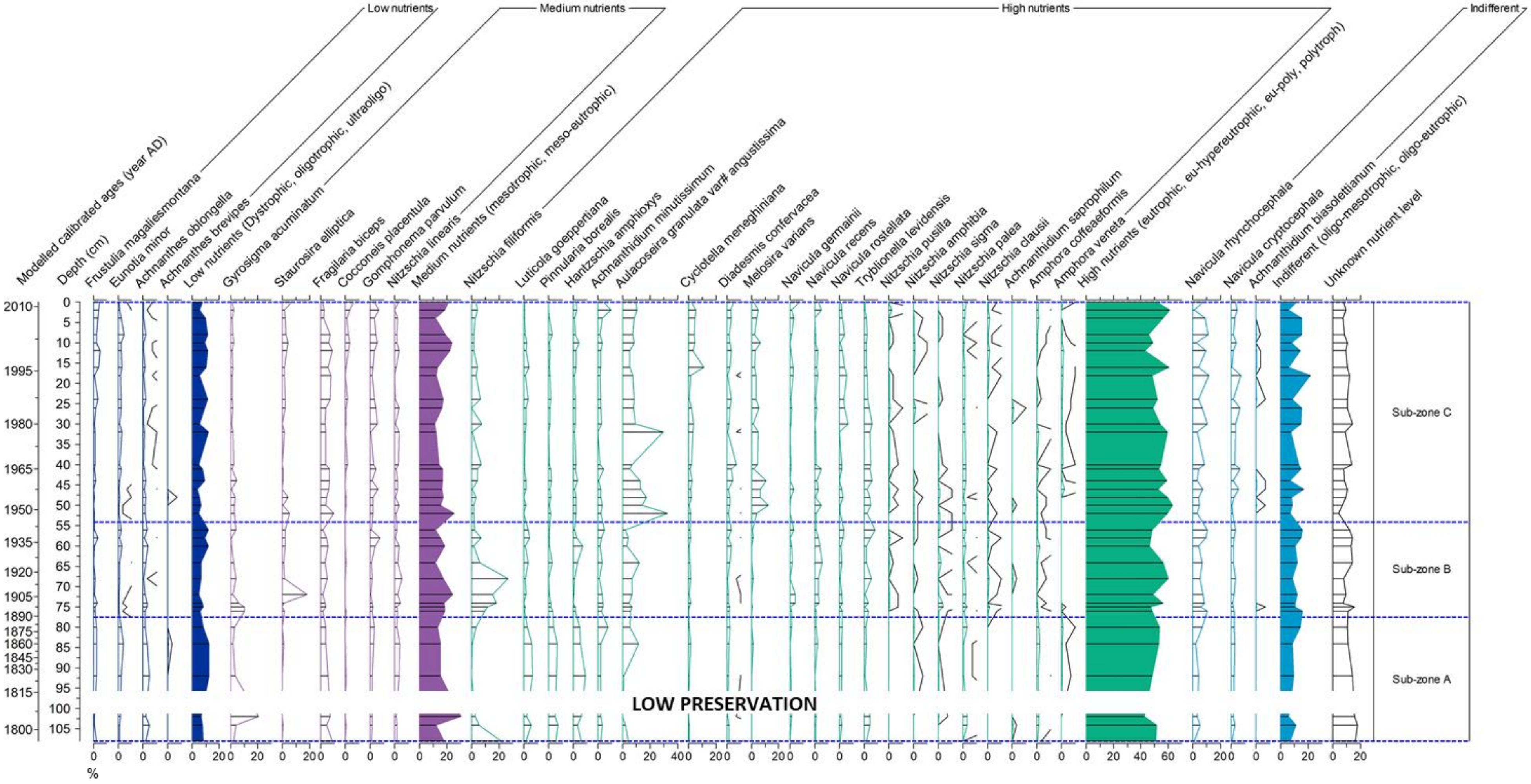



*Figure 10.21: Percentage diatom diagram with selected diatom taxa that depict nutrient preference from the RV3 core at Rhenostervlei Farm, Middle Berg River Catchment, South Africa.. x10 exaggeration is shown since percentage values are often low for most taxa. Diatom taxa are grouped into nutrient preferences (low, medium, high and indifferent). Although not statistically significant, sub-zones (A, B, C) are indicated by the dashed lines. Modelled calibrated ages are shown on the far left of the diagram. Low diatom preservation is shown between 96-100 cm in Sub-zone A*

### 10.6.6 Appendix: NMDS ordination stress values and plot

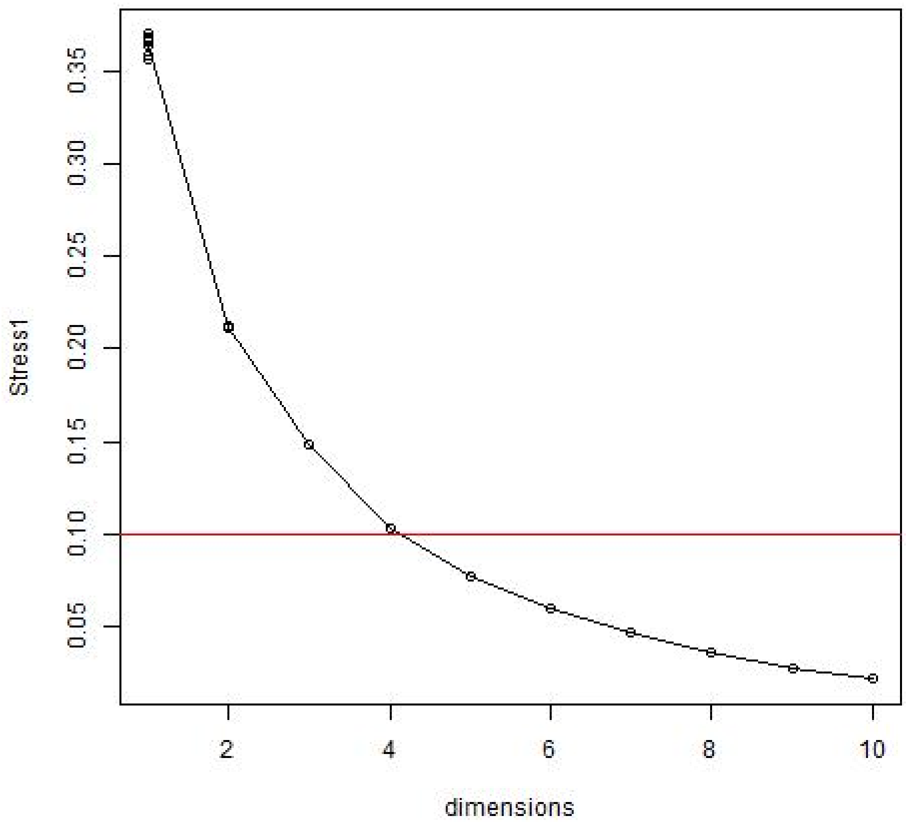



***Figure 10.22: Stress values for the NMDS ordination of the RV3 core. The number of dimensions/iterations was 4.***

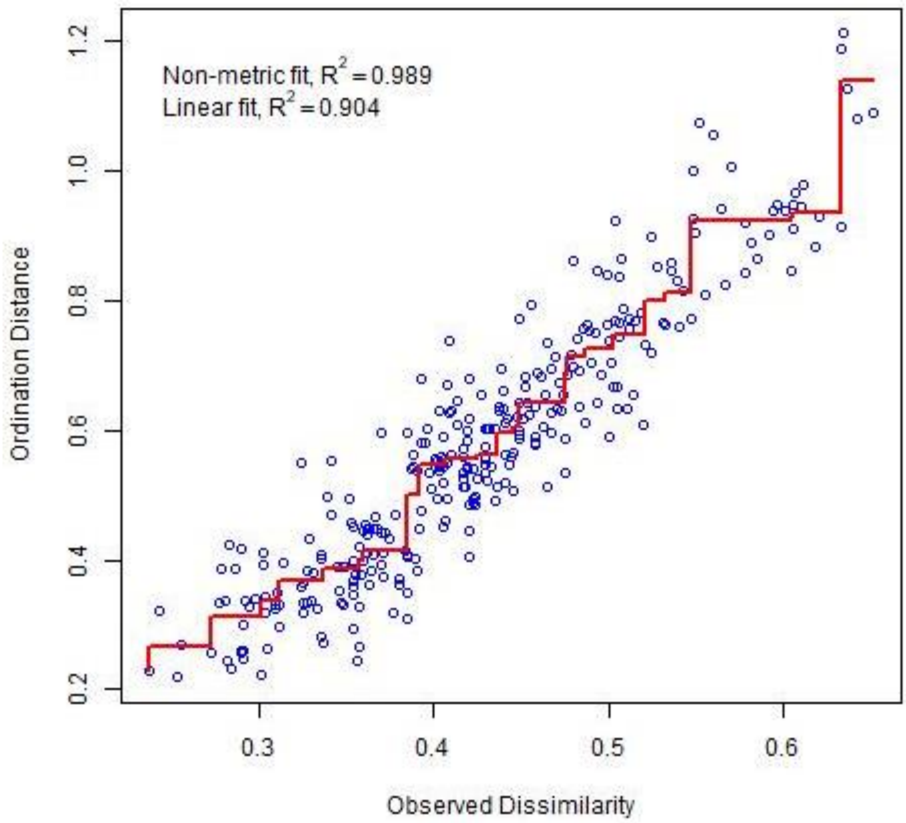



***Figure 10.23: Stress plot for the NMDS ordination of the RV3 core. Linear fit $R^2$ 0.9.***

## 10.7 Detailed results of the proof of concept for blending palaeoecology and PSD

### 10.7.1 Stakeholder engagement to inform SES problem articulation

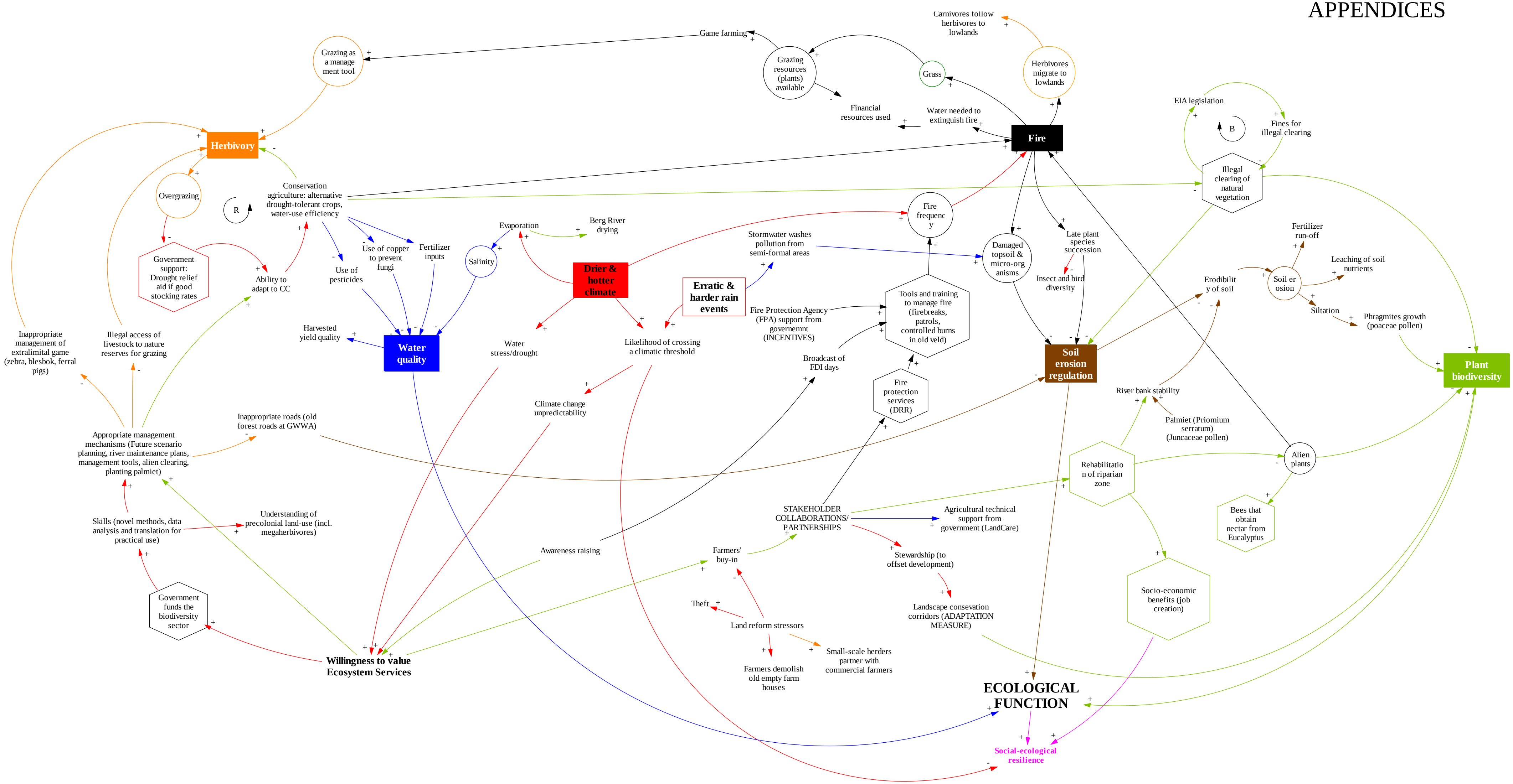



*Figure 10.24: Systems map of sustainable land-use management within the CFR generated through dialogue between multiple land-use managers during the multi-stakeholder engagement workshop. Dialogues were centred around selected ecosystem services (plant biodiversity, water quality and soil erosion regulation) and drivers of change (climate, herbivory and fire).*

## 10.7.2 Qualitative model (CLD) documentation

| Variable in CLD | Documentation |
|---|---|
| NAN(Success_of_palatable_Asteraceous_shrubs;Success_of_unpalatable_Renosterbos) | In the success to the successful archetype (competitive exclusion principle - Levin 1970, Weiner 2019) there is a limited pool of resources - e..g. space at EPNR, especially when land-use disturbance is absent (more time without disturbance). With increased grazing Renosterbos starts to become more successful even though historically (pre-1950s) it was not, it tends to garner more resources ( since it is unpalatable it is not grazed, and therefore is allocated more space after disturbance – e.g. fire). |
| NAN(Desired_level_of_plant_biodiversity) | Management plans that are informed by long-term data, systems-thinking and the social-ecological context. |
| Fire | Although fires can occur under a wide range of weather conditions, large fires are often restricted to periods of high fire danger, with increased climate change the likelihood of high FDI increases (Forsyth and van Wilgen 2008). Data used to calculate the fire danger index include daily maximum temperature and minimum relative humidity, wind speed, time since the last rain fell and moisture deficit (Noble et al. 1980).<br>Climate change also indirectly affects grazing resources (i.e. less rainfall directly affects primary productivity and therefore vegetation cover). However, it was a boundary decision to focus on grazing and fire as the main causes of biodiversity loss so the impacts of climate change on vegetation growth and grazing is not modelled. |
| NAN(Success_of_unpalatable_Renosterbos) | A pre-1950s or pre-colonial baseline of less Renosterbos (Forbes et al. 2018) |
| NAN("Appropriate_land-use_management";Climate_change;Renosterbos_homogenization) | Study at EPNR: "Burning reduced shrub cover and increased overall species richness and diversity. Burning also reduced grass biomass, and increased recruitment of indigenous seedlings." (Midoko-Iponga 2004)<br><br>Hot, dry weather together with the increased likelihood of fire ignition during the 21st century (associated with increased population) could increase fire frequency and intensity (Forsyth et al. 2000).<br>"On average one to three fires have occurred every decade since 1980. Local records suggest that a fire occurred in 1980; a relatively extensive fire in 1982 and 1988; a widespread wind-driven fire in 1999; less extensive fires in 2008 and 2010; and the most recent fire in 2012 (Figure 8; Wooding pers. com. 2013). These were wild fires that were most likely caused by humans. Fires always started in the Bainskloof area (east of Elandsberg PNR) and with the prevailing south-easterly wind during summer months, the fire comes off the mountain and into the lowlands. These fires often take place during the festive season (December-January) as this is when many people go up the mountain (Wooding pers. com. 2013). Visual inspection of burnt skeletons of Proteaceae and Elytropappus rhinocerotis confirmed that the site had burnt recently (pers. obs. 2012)." (Forbes 2014) |
| NAN(Fire;"Survival_of_disturbance-resistant_plants") | "Herbivores should be attracted to recently burned veld due to the higher quality of forage appearing in the more nutrient-rich, post-fire areas (Beukes 1987; Hobbs 1996; Archi- bald et al. 2005; Kraaij & Novellie 2010)."<br>Grazing lawn is desirable for herbivores but at this site grass pollen was relatively low over the ca. 1300 year period (Forbes et al. 2018). |
| NAN(Herbivores) | The more herbivores there are the more grazing will occur. |
| NAN(Palatable_plants_available_for_LIHs;"Livestock_agriculture_(1650s-1970s)";"Large_indigenous_herbivores_reintroduced_(1970s-present)") | LSU data from Elandsberg PNR - 1994-2019<br>Estimated LSU data based on LSU data from Wellington District 1928-1995. Calculations and Assumptions:<br>1. average LSU/ha in the Wellington District was applied in EPNR (4000 ha) and;<br>2. Anecdotal evidence about more conservative livestock numbers at EPNR due to poorer quality veld therefore the LSU was further divided by two.<br><br>Pers. Comm. Wooding 2020: "I have no other game figures or livestock figures from before 1994. Until the 1980's the farm "kept sheep and cattle on the veld camps (now nature reserve)and fed them in the summer". One thing we have been discussing is reducing the game numbers over drought periods but even then it would be to 200 LSU rather than 250 LSU, so not to a dramatic change."<br><br>Timeline describing long-term climatic, vegetation and land-use change during the study period (ca. 1300 years) documented by the VANG core. Inserts highlighted in bold are specific to Elandsberg PNR. (Table 1 in Forbes 2014) |
| NAN("Appropriate_land-use_management") | Coprophilous fungal spore data from Forbes et al. 2018. |
| NAN(Success_of_unpalatable_Renosterbos;Success_of_palatable_Asteraceous_shrubs;Grass) | Palatable plant biomass available for large indigenous herbivores (LIH) would increase the the LSU. |
| NAN(Allocation_of_space_to_R_instead_of_A_shrubs) | Since a patch-mosaic landscape if preferable and the variety in vegetation units can act as natural fire refugia (e.g. thicket by boulders) (Forbes et al. 2018). The timing of fires (i.e. burn season in autumn versus spring; Levyns 1935; Cowling et al. 1986) and grazing (i.e. immediately after a burn or not; Rebelo 1995) are important factors that will effect biodiversity and patch structure and therefore burn differently when a fire comes through. Assumption is the effect is a S-shaped decrease. |
| NAN(Allocation_of_space_to_R_instead_of_A_sh | More resources go to increasing other Asteraceaous shrubs cover. More resources go to increasing Renosterbos cover. |
| NAN(Resources_to_Asteraceous_shrubs;"Survival_of_disturbance-resistant_plants";Fire) | Study at EPNR: "Experimental transplanting of indigenous shrubs into an old field showed that most of the plants investigated competed for resources with lawn grasses on the field, and competition affected the seedlings throughout the experiment. Mortality was higher, and growth was reduced for seedlings exposed to grass competition." (Midoko-Iponga 2004) "grass can reduce recruitment and growth of many indigenous shrub species." (Midoko-Iponga 2004) |
| NAN(Resources_to_Renosterbos;Fire) | Radloff et al. 2014 found (see pg 10/14):"The burned shrubland plots did not convert to grazing lawn in the presence of herbivory, but returned to a nanophyllous shrubland state. This was not due to the resprouting of E. rhinocerotis from epicormic buds of roots (we did not observe a single case of such resprouting in the burned area), but from the successful germination of seedlings. Failure to regenerate from epicormic buds may be due to the severity of the fire that could have killed all the original shrubs (C. Boucher, pers. com., Dec 2011)." |
| NAN(Grazing) | "It is possible that even unpalatable shrubs – such as the common Elytropappus rhinocerotis – will be removed just after they have germinated as a consequence of intensive and indiscriminate trampling and grazing by grazers (Hester et al. 2000) or through deliberate selection by browsers (Scholes & Archer 1997; Augustine & McNaughton 1998)."<br><br>Study at EPNR showed that "Experimental transplanting of indigenous shrubs into an old field showed that most of the plants investigated competed for resources with lawn grasses on the field, and competition affected the seedlings throughout the experiment. Mortality was higher, and growth was reduced for seedlings exposed to grass competition." (Midoko-Iponga 2004)<br><br>"If the burnt area in this generally nutrient-poor environment (Joubert & Stindt 1979) is small, both browsers and grazers will gather in the post-fire recovering habitats in such numbers that only the most grazing- and trampling-resistant plant species would survive the intense disturbance, consequently leading to the formation of grazing lawns" (Hobbs 1996; Archibald et al. 2005).<br><br>Radloff et al. 2014, "E. rhinocerotis shrub seedlings are clearly unpalatable and sensitive to competition, as they were not removed by the mix-feeding eland or ostrich in the grazed plots, and failed to establish in high numbers in the presence of competing shrubs and grasses within the exclosures." |

### 10.7.3 Large Stock Unit (LSU) data from Elandsberg PNR

***Table 10.8: Historical Large Stock Unit (LSU) data (1994-2019) for Elandsberg PNR***(Wooding, 2019, 2018, 2013)**.**

| | | | 1994 | 1996 | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 | 2008 | 2010 | 2011 | 2012 | 2013 | 2014 | 2018 | 2019 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Species** | | **LSU Value for game species** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** | **LSU** |
| Eland | Mixed | 1.02 | 137 | 87 | 91.8 | 92 | 107.1 | 122 | 108.12 | 116 | 112.2 | 149 | 134.64 | 95.88 | 116.28 | 113.22 | 122 | 79 | 100 |
| Bontebok | Mixed | 0.22 | 48 | 22 | 22 | 22 | 28.6 | 37 | 33 | 34 | 27.06 | 22 | 24.2 | 18.7 | 19.36 | 20.68 | 26 | 18 | 23 |
| Springbok | Mixed | 0.15 | 37 | 16 | 16.5 | 18 | 24 | 30 | 38.25 | 47 | 37.5 | 30 | 31.95 | 30 | 43.2 | 43.05 | 30 | 62 | 53 |
| Zebra | Mixed | 0.66 | 0 | 23 | 18.48 | 20 | 19.8 | 20 | 18.48 | 20 | 19.8 | 13 | 14.52 | 19.14 | 19.8 | 13.86 | 20 | 17 | 15 |
| Hartebeest | Grazer | 0.37 | 20 | 10 | 5.18 | 4 | 5.18 | 5 | 7.4 | 9 | 7.4 | 3 | 6.66 | 9.25 | 9.99 | 10.73 | 11 | 13 | 11 |
| Mnt Zebra | Grazer | 0.62 | 0 | 0 | 2.48 | 0 | 2.48 | 0 | 3.1 | 3 | 3.72 | 4 | 4.34 | 5.58 | 4.96 | 4.96 | | 6 | 7 |
| Ostrich | Grazer | 0.25 | 0 | 12 | 11.5 | 11 | 9.75 | 8 | 6.25 | 5 | 4.75 | 5 | 4 | 5 | 5.5 | 3.25 | 7 | 4 | 4 |
| Gemsbok | Grazer | 0.56 | 20 | 13 | 7.84 | 6 | 6.72 | 7 | 7.28 | 8 | 8.4 | 15 | 10.64 | 12.88 | 14 | 19.04 | 14 | 21 | 23 |
| Black Wilde | Grazer | 0.46 | 16 | 13 | 15.64 | 18 | 22.54 | 27 | 29.44 | 38 | 19.32 | 29 | 14.72 | 9.66 | 12.88 | 13.8 | 18 | 9 | 14 |
| Grey Rhebok | Grazer | 0.1 | 0 | 0 | 1.4 | 0 | 1.8 | 0 | 1.2 | 0 | 1.4 | 0 | 1.2 | 1.5 | 0.5 | 0.7 | 3 | 1 | 1 |
| Blue Wildeb | Grazer | 0.46 | 34 | 21 | 33.8 | 35 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | **Total LSU per yr** | **312** | **216** | **227** | **226** | **228** | **256** | **253** | **280** | **242** | **268** | **247** | **208** | **246** | **243** | **251** | **228** | **251** |

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### 10.7.4 Ecological Model structure and documentation

***Table 10.9: Detailed model documentation for the Ecological Model for Elandsberg PNR. The Stella Architect model files for the Ecological Mode can be found via the following google drive link: https://drive.google.com/file/d/1VrBckYm3dsFte7Vy_PNpnxUFRyKV4j4T/view?usp=sharing***

| Total | Count |
|---|---|
| Variables | 92 |
| Modules | 1 |
| Sectors | 4 |
| Stocks | 3 |
| Flows | 6 |
| Converters | 83 |
| Constants | 48 |
| Equations | 41 |
| Graphicals | 8 |
| **Run Specs** | |
| Start Time | 750 |
| Stop Time | 2100 |
| DT | 0,25 |
| Fractional DT | FALSE |
| Save Interval | 0,25 |
| Sim Duration | 1,49883809 |
| Time Units | Year |
| Pause Interval | 0 |
| Integration Method | Euler |
| Keep all variable results | TRUE |
| Run By | Run |
| Calculate loop dominance information | FALSE |

| Variable | Equation | Units |
|---|---|---|
| Data:_Asteraceae_pollen | GRAPH(TIME) Points: (751, 0,20233463), (876, 0,220883534), (1001, 0,20610687), (1124, 0,274900398), (1189, 0,298804781), (1234, 0,416), (1267, 0,417808219), (1276, 0,37007874), (1285, 0,444444444), (1291, 0,432539683), (1302, 0,502040816), (1323, 0,473684211), (1374, 0,416666667), (1438, 0,509881423), (1527, 0,442687747), (1671, 0,485714286), (1779, 0,564705882), (1895, 0,405511811), (1943, 0,367924528), (1958, 0,359504132), (1959, 0,281746032), (1961, 0,283464567), (1962, 0,266932271), (1967, 0,302325581), (1977, 0,220532319), (1981, 0,245033113), (1986, 0,215686275), (1989, 0,211320755), (1994, 0,176029963), (1997, 0,220973783), (2000, 0,130769231), (2004, 0,16875), (2005, 0,142322097), (2006, 0,133858268), (2009, 0,115942029), (2012, 0,155378486) | 1 |
| "Data:_Macro-charcoal_(local_fire)" | GRAPH(TIME) Points: (751, 44), (876, 268), (1001, 50), (1124, 74), (1189, 164), (1234, 116), (1267, 46), (1276, 170), (1285, 110), (1291, 140), (1302, 174), (1323, 86), (1374, 100), (1438, 202), (1527, 184), (1671, 254), (1779, 164), (1895, 256), (1943, 158), (1958, 400), (1959, 324), (1961, 190), (1962, 530), (1967, 432), (1977, 450), (1981, 352), (1986, 434), (1989, 500), (1994, 446), (1997, 676), (2000, 410), (2004, 820), (2005, 1100), (2006, 902), (2009, 662), (2012, 160) | n cm^-3 |
| Data:_Renosterbos_pollen | GRAPH(TIME) Points: (751, 0,13229572), (876, 0,200803213), (1001, 0,179389313), (1124, 0,151394422), (1189, 0,215139442), (1234, 0,160), (1267, 0,195205479), (1276, 0,149606299), (1285, 0,182539683), (1291, 0,202380952), (1302, 0,191836735), (1323, 0,135338346), (1374, 0,200617284), (1438, 0,201581028), (1527, 0,14229249), (1671, 0,228571429), (1779, 0,184313725), (1895, 0,25984252), (1943, 0,182389937), (1958, 0,400793651), (1959, 0,409448819), (1961, 0,301652893), (1962, 0,422310757), (1967, 0,360465116), (1977, 0,448669202), (1981, 0,490066225), (1986, 0,447058824), (1989, 0,535849057), (1994, 0,558052434), (1997, 0,546816479), (2000, 0,607692308), (2004, 0,478125), (2005, 0,610486891), (2006, 0,570866142), (2009, 0,597826087), (2012, 0,609561753) | 1 |

| Variable | Equation | Units | Documentation |
|---|---|---|---|
| **"Plant_biodiversity_(provisioning/supporting_ecosystem_service)": Asteraceous shrubs are an indicator of higher plant biodiversity (ecosystem service) than the Renosterbos shrub (which is a single species and unpalatable therefore not desired by herbivores).** | | | |
| Asteraceous_pollen_relative_decrease_rate | 0,0159 {0.0892818135101} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. There was no need to have different rates for pre- and post-1800s in this case since the optimization revealed similar rates. |
| Asteraceous_shrubs(t) | Asteraceous_shrubs(t - dt) + (Increase_in_Asteraceous_pollen - Decrease_in_Asteraceous_pollen) * dt | 1 | Initial value is 0.2 as per data from Forbes et al. (2018). Unit is dimensionless since it is pollen proportion (percentage) data. |
| Decrease_in_Asteraceous_pollen | Asteraceous_shrubs*Actual_effect_of_grazing*Actual_effect_of_fire_on_A*Asteraceous_pollen_relative_decrease_rate | 1/Year | This is the amount of Asteraceous pollen that decreases each year. |
| Decrease_in_Renosterbos_pollen | IF TIME <1800 THEN "Pre-1800s_Renostebos_pollen_relative_decrease_rate"*Renosterbos*Actual_effect_of_fire_on_Renosterbos ELSE "Post-1800s_Renostebos_pollen_relative_decrease_rate"*Renosterbos*Actual_effect_of_fire_on_Renosterbos | 1/Year | This is the amount of Renosterbos pollen that decreases each year. |
| Max_Renosterbos | 0,61 | 1 | This is the maximum observed Renosterbos pollen in the palaeo record from this site (Forbes et al.2018). It is % or proportion data. |
| FRED_pollen | 1-Total_pollen_abundance/Max_total_pollen_abundance | 1 | Assumption: This is the pollen reduction factor that relates to carrying capacity or competition. Competition between organisms that are using the same fuel resources or space/shelter that become in short supply will lead to reduction in population growth (Harvey et al. 2008).<br>The Logistic model sets limit to the growth using this equation: (dN/dt)=r*N*(1-(N/K)). The logistic differential equation, has N as the population size, r is growth rate, K is carrying capacity. This equation forces, populations to converge to the carrying capacity. The speed at which the populations approach K is related to the growth rate r. |
| Increase_in_Asteraceous_pollen | IF TIME <1800 THEN "Pre-1800s_Asteraceous_shrubs_pollen_relative_increase_rate"*Asteraceous_shrubs*FRED_pollen ELSE "Post-1800s_Asteraceous_shrubs_pollen_relative_increase_rate"*Asteraceous_shrubs*FRED_pollen | 1/Year | This is the amount of Asteraceous pollen that increases each year. |
| Increase_in_Renosterbos_pollen | IF TIME<1800 THEN Renosterbos*"Pre-1800s_Renostebos_pollen_relative_increase_rate"*FRED_pollen ELSE "Post-1800s_Renostebos_pollen_relative_increase_rate"*FRED_pollen | 1/Year | This is the amount of Renosterbos pollen that increases each year. |
| Max_total_pollen_abundance | 0,77 | 1 | Calculated: This is the maximum total abundance of Asteraceous shrubs pollen plus Renosterbos pollen (77%) in the palaeoecological record. The remainder (33%) is made up of other pollen taxa not included in this model (Forbes et al. 2018). |
| "Post-1800s_Asteraceous_shrubs_pollen_relative_increase_rate" | 0,0352 {0.0861878866808} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Post-1800s_Renostebos_pollen_relative_decrease_rate" | 0,2407 {0.254476344716} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Post-1800s_Renostebos_pollen_relative_increase_rate" | 0,204689114943 {0.204453075701} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Pre-1800s_Asteraceous_shrubs_pollen_relative_increase_rate" | 0,0031 {0.00340569861543} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Pre-1800s_Renostebos_pollen_relative_decrease_rate" | 0,2340 {0.256609081612} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Pre-1800s_Renostebos_pollen_relative_increase_rate" | 0,0225 {0.00682718355723} | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| Relative_abundance_in_Asteraceous_pollen | (MIN(Asteraceous_shrubs/Renosterbos; 1)) | 1 | This is comparing the relative abundance of Asteraceous shrubs and Renosterbos to determine whether there is a well-balanced ratio or if Renosterbos is dominant and therefore having a homogenization effect. |
| Renosterbos(t) | Renosterbos(t - dt) + (Increase_in_Renosterbos_pollen - Decrease_in_Renosterbos_pollen) * dt | 1 | Initial value is 0.13 as per data from Forbes et al. (2018). Unit is dimensionless since it is pollen proportion (percentage) data. |
| Total_pollen_abundance | Renosterbos+Asteraceous_shrubs | 1 | This is the observed total Asteraceous and Renosterbos pollen. |

| Drivers_of_change: Fire and grazing are considered as land-use drivers of change. However, fire is also influenced by climate (hot dry conditions). | | | |
|---|---|---|---|
| Actual_effect_of_fire_on_A | Effect_of_fire_on_decrease_in_Astreraceous_pollen*Standard_effect_of_fire_on_Asteraceous | 1 | |
| Actual_effect_of_fire_on_Renosterbos | Effect_of_fire_on_decrease_in_Renosterbos_pollen*Standard_effect_of_fire_on_Renosterbos | 1 | Assumption: The standard effect can be calculated using the palaeo data to optimize for the standard effect. The standard effect is then multiplied by the assumed effect that fire has on decreasing Renosterbos. |
| Actual_effect_of_grazing | Effect_of_grazing_on_decreasing_Asteraceous_pollen*Standard_effect_of_grazing | 1 | Assumption: The standard effect can be calculated using the palaeo data to optimize for the standard effect of grazing. The standard effect is then multiplied by the assumed effect that grazing has on decreasing Asteraceous shrubs since they are palatable and Renosterbos is not. |
| Actual_effect_of_homogenous_Renosterbos_on_fire | Effect_of_Renosterbos_homogenization_on_fire*Standard_effect_of_homogenization_on_fire | 1 | The actual effect is based on the standard effect (calculated via optimization in Stella) multiplied by the assumed effect of homogenization. |
| Decreasing_fire | IF TIME <1800 THEN Fire*"Pre-1800s_fire_relative_decrease_rate" ELSE Fire*"Post-1800s_fire_relative_decrease_rate" | n cm^-3/Year | This is the amount of macro-charcoal that decreases each year. |
| Effect_of_fire_on_decrease_in_Astreraceous_pollen | GRAPH(Fire/Potential_max_fire) Points: (0,000, 0,000), (0,100, 0,0612070245601), (0,200, 0,128851248086), (0,300, 0,203609676702), (0,400, 0,28623051789), (0,500, 0,377540668798), (0,600, 0,478453992107), (0,700, 0,589980462274), (0,800, | 1 | Assumption that the effect will be that of exponential growth. Renosterveld and Fynbos taxa are fire-prone and fire-adapted (Bond 1980; van Wilgen. 1982; van Wilgen et al. 2012) |
| Effect_of_fire_on_decrease_in_Renosterbos_pollen | GRAPH(Fire/Potential_max_fire) Points: (0,000, 0,000), (0,100, 0,0612070245601), (0,200, 0,128851248086), (0,300, 0,203609676702), (0,400, 0,28623051789), (0,500, 0,377540668798), (0,600, 0,478453992107), (0,700, 0,589980462274), (0,800, 0,713236273698), (0,900, 0,849455011967), (1,000, 1,000) | 1 | Assumption that the effect will be that of exponential growth. Renosterveld and Fynbos taxa are fire-prone and fire-adapted (Bond 1980; van Wilgen. 1982; van Wilgen et al. 2012). |
| Effect_of_grazing_on_decreasing_Asteraceous_pollen | GRAPH(Grazing/Max_grazing) Points: (0,000, 0,000), (0,100, 0,33583091167), (0,200, 0,560945103841), (0,300, 0,7118436595), (0,400, 0,812993986277), (0,500, 0,880797077978), (0,600, 0,926246849528), (0,700, 0,956712742486), (0,800, 0,977134641257), (0,900, 0,99082384938), (1,000, 1,000) | 1 | Asteraceous shrubs decrease since they are palatable for herbivores. Assumption that there it is an exponential increase since the more resources are available, the more grazing will occur and increased herbivore population. The herbivore population will eventually reach a carrying capacity. |
| Effect_of_Renosterbos_homogenization_on_fire | GRAPH(Relative_abundance_in_Asteraceous_pollen*Max_Renosterbos) Points: (0,000, 0,993307149076), (0,100, 0,982013790038), (0,200, 0,952574126822), (0,300, 0,880797077978), (0,400, 0,73105857863), (0,500, 0,500), (0,600, 0,26894142137), (0,700, 0,119202922022), (0,800, 0,0474258731776), (0,900, 0,0179862099621), (1,000, 0,00669285092428) | 1 | Since a patch-mosaic landscape if preferable and the variety in vegetation units can act as natural fire refugia (e.g. thicket by boulders) (Forbes et al. 2018). The timing of fires (i.e. burn season in autumn versus spring; Levyns 1935; Cowling et al. 1986) and grazing (i.e. immediately after a burn or not; Rebelo 1995) are important factors that will effect biodiversity and patch structure and therefore burn differently when a fire comes through. Assumption is the effect is a S-shaped decrease. |
| Fire(t) | Fire(t - dt) + (Increasing_fire - Decreasing_fire) * dt | n cm^-3 | Initial value is 74 as per data from Forbes et al. (2018). Unit is number of macro-charcoal particles per cubic cm. |
| FRED_Fire | 1-(Fire/Potential_max_fire) | 1 | Assumption: This is the assumed reduction factor for fire.<br>Similarly to the population growth equation used for the pollen, here we set a limit to the growth of the fire as the fuel load is depleted over time since the area carries a finite amount of fuel.<br>(dN/dt)=r*N*(1-(N/K)). The logistic differential equation, has N as the population size, r is growth rate, K is carrying capacity. |
| Increasing_fire | IF TIME <1800 THEN Fire*"Pre-1800s_fire_relative_increase_rate"*Actual_effect_of_homogenous_Renosterbos_on_fire*FRED_Fire ELSE Fire*"Post-1800s_fire_relative_increase_rate"*Actual_effect_of_homogenous_Renosterbos_on_fire*FRED_Fire | n cm^-3/Year | This is the amount of macro-charcoal that increases each year. |
| Max_grazing | 0,86 | 1 | This is the maximum observed dung fungal spores in the palaeo record from this site (Forbes et al.2018). It is % or proportion data. |
| Potential_max_fire | 2200 | n cm^-3 | Assumption that potential max fire can be double (2200 n cm^-3) the observed max fire (1100 n cm^-3). This was also optimized in Stella. |
| "Pre-1800s_fire_relative_decrease_rate" | 0,1009 {0.0789060598405} | 1/Year | Calculated: Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Pre-1800s_fire_relative_increase_rate" | 0,3636 {0.307658554639} | 1/Year | Calculated: Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| Standard_effect_of_fire_on_Asteraceous | 1,0249 {1.03382231776} | 1 | Calculated using optimization in Stella: Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. |
| Standard_effect_of_fire_on_Renosterbos | 0,9998 | 1 | Calculated using optimization in Stella: Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. |
| Standard_effect_of_grazing | 2,6455 {2.57594738462} | 1 | Calculated using optimization in Stella: Based on reliable dung fungal spores data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. |
| Standard_effect_of_homogenization_on_fire | 1,1432 {1.08187668109} | 1 | Calculated using optimization in Stella: Based on reliable palaeo data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. |

| POLICIES_FOR_RESERVE_MANAGERS: | | | |
|---|---|---|---|
| "NEW_SWITCH_P1._2" | 0 | 1/Year | New switch created so that I could use these variables in Policy 3 which is a combination of Policy 1.1 and 1.2 and Policy 2. Or use it separately without switching on Policy 3. This switch turns on Policy 1.2. which increases the fire decrease rate due to improved fire control. increased fire control via more pro-active Fire Protection Associations - e.g. law enforcement of firebreaks (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "NEW_SWITCH_P1.1" | 0 | 1/Year | New switch created so that I could use these variables in Policy 3 which is a combination of Policy 1.1 and 1.2 and Policy 2. Or use it separately without switching on Policy 3. This switch turns on Policy 1.1. which decreases the fire increase rate due to increased fire prevention in the form of public awareness raising about high Fire Danger Index days (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "POLICY_1.1:_FIRE_PREVENTION_TO_DECREASE_FIRE" | STEP(Fire_prevention:_Decrease_fire_increase_rate; 2020) | 1/Year | Stakeholders' suggestion to prevent fire from taking place by having awareness-raising and fire alerts (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "POLICY_1.2:_FIRE_CONTROL_TO_DECREASE_FIRE" | STEP(Fire_control:_Increase_fire_decrease_rate; 2020) | 1/Year | Policy 1.2. which increases the fire decrease rate due to improved fire control. increased fire control via more pro-active Fire Protection Associations - e.g. law enforcement of firebreaks (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "SWITCH_POLICY_1.1" | (1-"NEW_SWITCH_P1.1")*(0+"POLICY_3:_ADAPTIVE_FIRE-GRAZING_MANAGEMENT")+("NEW_SWITCH_P1.1")*1 | 1/Year | Policy 1.1. which decreases the fire increase rate due to increased fire prevention in the form of public awareness raising about high Fire Danger Index days (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "SWITCH_POLICY_1.2" | (1-"NEW_SWITCH_P1._2")*(0+"POLICY_3:_ADAPTIVE_FIRE-GRAZING_MANAGEMENT")+("NEW_SWITCH_P1._2")*1 | 1/Year | Policy 1.2. which increases the fire decrease rate due to improved fire control. increased fire control via more pro-active Fire Protection Associations - e.g. law enforcement of firebreaks (Multi-stakeholder social learning workshop 1 - 4 July 2019). |
| "2012_level_of_grazing" | 0,531914893 | 1 | 2012 level of dung fungal spores is 53% (Forbes et al. 2018) |
| Change_grazing_back_up | ((1-SWITCH_POLICY_2)*0+(SWITCH_POLICY_2)*(STEP("Size_of_change_(actual_change_in_grazing)";2020+Duration_of_policy:Time_to_adjust_grazing_management))) | 1 | Return grazing to what it was previously. |
| Change_grazing_down | ((1-SWITCH_POLICY_2)*0+(SWITCH_POLICY_2)*(STEP("Size_of_change_(actual_change_in_grazing)"; 2020))) | 1 | The safe operating space is between zero grazing to the max level of grazing before the regime shift in 1950s, therefore 0-31%. Ideally grazing would be at the pre-1950s average of 9.35%. I have provided the change here, thus 2012 level (53%) minus desired level (9.35%): 53-9.35 = 43.65% (0.4356). Policy sensitivity needs to be done to determine how long this policy needs to be in place before managers can increase the stock again. |
| "Data:_Coprophilous_fungal_spores_(grazing)" | GRAPH(TIME) Points: (751, 0,03125), (876, 0,000), (1001, 0,000), (1124, 0,100), (1189, 0,000), (1234, 0,086956522), (1267, 0,000), (1276, 0,000), (1285, 0,020), (1291, 0,011494253), (1302, 0,057692307), (1323, 0,140), (1374, 0,02173913), (1438, 0,105263158), (1527, 0,189189189), (1671, 0,175), (1779, 0,229007634), (1895, 0,296296296), (1943, 0,312499999), (1958, 0,362318841), (1959, 0,418410043), (1961, 0,448430494), (1962, 0,460829494), (1967, 0,4784689), (1977, 0,655502393), (1981, 0,612121211), (1986, 0,731884057), (1989, 0,775193799), (1994, 0,75147929), (1997, 0,812499999), (2000, 0,869257951), (2004, 0,835390947), (2005, 0,834821429), (2006, | 1 | Grazing is currently an exogenous variable. Dung fungal spore data from 751 to 2012 (Forbes et al. 2018). |
| Duration_of_policy:Time_to_adjust_grazing_management | 50 | Year | This is the duration of time while the grazing reduction policy is in place before it can return to its previous level. |
| Fire_control:_Increase_fire_decrease_rate | (1-"SWITCH_POLICY_1.2")*0+("SWITCH_POLICY_1.2")*0,1 | 1/Year | Stakeholders' suggestion to improve fire control have enforcing fire break laws and improving Fire Support Association initiatives such as training during controlled burns (Reference: SE Workshop 1). Assumption: 0.1 rate to decrease fire per year. |
| Fire_prevention:_Decrease_fire_increase_rate | (1-"SWITCH_POLICY_1.1")*0+("SWITCH_POLICY_1.1")*0,1 | 1/Year | Stakeholders' suggestion to prevent fire from taking place by having awareness-raising and fire alerts (Multi-stakeholder social learning workshop 1 - 4 July 2019). Assumption: 0.1 rate to decrease fire per year. |
| Grazing | IF TIME <=2012 THEN "Data:_Coprophilous_fungal_spores_(grazing)" ELSE Grazing_after_2012 | 1 | This is the palaeo data from 751-2012 plus the new modelled data associated with the policy from 2012-2100. |
| Grazing_after_2012 | "2012_level_of_grazing"-POLICY_2:_DECREASE_GRAZING+Change_grazing_back_up | | This is the level of grazing after 2012 until 2100. |
| POLICY_2:_DECREASE_GRAZING | STEP(Change_grazing_down; 2020) | 1 | Policy 2 is the decision taken by Elandsberg PNR to reduce their stocking number to the pre-1950s level (9.35%) (Forbes et al. 2018) for certain period of time (5, 10 or 50 years). |
| "POLICY_3:_ADAPTIVE_FIRE-GRAZING_MANAGEMENT" | (1-SWITCH_POLICY_3)*STEP(0; 2012+Duration_of_policy:Time_to_adjust_grazing_management)+(SWITCH_POLICY_3)*STEP(1; 2012+Duration_of_policy:Time_to_adjust_grazing_management) | 1 | Policy 3 is Adaptive grazing-fire management where Elandsberg PNR managers decrease grazing to the average pre-1950s levels (9.35%) for a certain period of time (5, 10 or 50 years) and then also implement management that aims to decrease fire (via fire prevention - Policy 1.1 and control - Policy 1.2) |
| "Post-1800s_fire_relative_decrease_rate" | 0,0517+"POLICY_1.2:_FIRE_CONTROL_TO_DECREASE_FIRE" {0.0468295779332} | 1/Year | Calculated: Based on reliable macro-charcoal data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Post-1800s_fire_relative_increase_rate" | 0,2292-"POLICY_1.1:_FIRE_PREVENTION_TO_DECREASE_FIRE"+SCENARIO_2:_CLIMATE_WARMING+"\"SCENARIO_1:_10_yr_CAPE_DROUGHT\"" | 1/Year | Calculated: Based on reliable palaeo-data collection (Forbes et al. 2018), and relatively sound model structure, optimization in Stella was used to determine the appropriate rate. Since a regime shift was detected in the palaeo-record, it was determined that the rates of change were different pre- and post-1800s. |
| "Size_of_change_(actual_change_in_grazing)" | 0,4356 {0.4356 for pre-1950s average of 9.35%} | 1 | The "current" (at 2012) level of grazing was 53%, therefore, can test a few values:<br>- 0.4356 means grazing will be reduced to 9.35% (i.e..53- 43.65 = 9.35, the average grazing level pre-1950s before the regime shift).<br>- 00.53 means grazing will be reduced to 0 (i.e. 53-53 = 0) (Forbes et al. 2018) |
| SWITCH_POLICY_2 | 0 | 1 | Switch that turns on Policy 2. |
| SWITCH_POLICY_3 | 0 | 1 | Switch that turns on Policy 3. Policy 3 is Adaptive grazing-fire management where Elandsberg PNR managers decrease grazing to the average pre-1950s levels (9.35%) for a certain period of time (5, 10 or 50 years) and then also implement management that aims to decrease fire (via fire prevention - Policy 1.1 and control - Policy 1.2) |

| FUTURE_CLIMATE_CHANGE_SCENARIOS: | | | |
|---|---|---|---|
| "\"SCENARIO_1:_10_yr_CAPE_DROUGHT\"" | Change_up-Change_down | n cm^-3/Year | |
| Change_down | ((1-SWITCH_DROUGHT_SCENARIO_1)*0+(SWITCH_DROUGHT_SCENARIO_1)*(STEP("Size_of_change_(actual_change_in_fire)"; 2015+Length_of_climate_effect))) | 1 | Return levels of fire to what it was before after the 10 year drought conditions. |
| Change_up | ((1-SWITCH_DROUGHT_SCENARIO_1)*0+(SWITCH_DROUGHT_SCENARIO_1)*(STEP("Size_of_change_(actual_change_in_fire)"; 2015))) | n cm^-3 | Increase fire to a higher level. This is an assumed amount. Additional data is required. |
| Length_of_climate_effect | 10 | Year | The current Cape drought is ongoing in various regions since 2015. This is the worst drought in 100 years. Assumption that dry conditions will persist for another 6 years before receiving improved average rainfall. Improved data needs to be obtained for this region to make this more realistic. |
| SCENARIO_2:_CLIMATE_WARMING | RAMP(Slope_of_increase_in_climate_change_affecting_fire; 2012) | n cm^-3/Year | Scenario 2 assumes that fires will increase at a certain steady rate after the year 2020. The rate is an assumption about how climate change increases the Fire Danger Index days and there are more fires due to hot and dry conditions (stakeholder workshop- July 2019) and van Wilgen et al. (2012). |
| "Size_of_change_(actual_change_in_fire)" | 0,1 | n cm^-3 | This size of change is an assumed rate. Additional data from Fynbos fire dynamics experts are required. |
| Slope | 0,001 | n cm^-3/Year | This is the slope of increase in fire after 2020. |
| Slope_of_increase_in_climate_change_affecting_fire | (0*(1-SWITCH_CLIMATE_WARMING_SCENARIO)+Slope*(SWITCH_CLIMATE_WARMING_SCENARIO)) | n cm^-3/Year | This slope assumes that climate change will cause fire to increase at a steady state from 2020. |
| SWITCH_CLIMATE_WARMING_SCENARIO | 0 | Year | Switch for Scenario 2 to increase fire due to cilmate change. |
| SWITCH_DROUGHT_SCENARIO_1 | 0 | n cm^-3 | Switch for Scenario 1 to increase fire due to cilmate change. |

### 10.7.5 Model behaviour sensitivity analysis

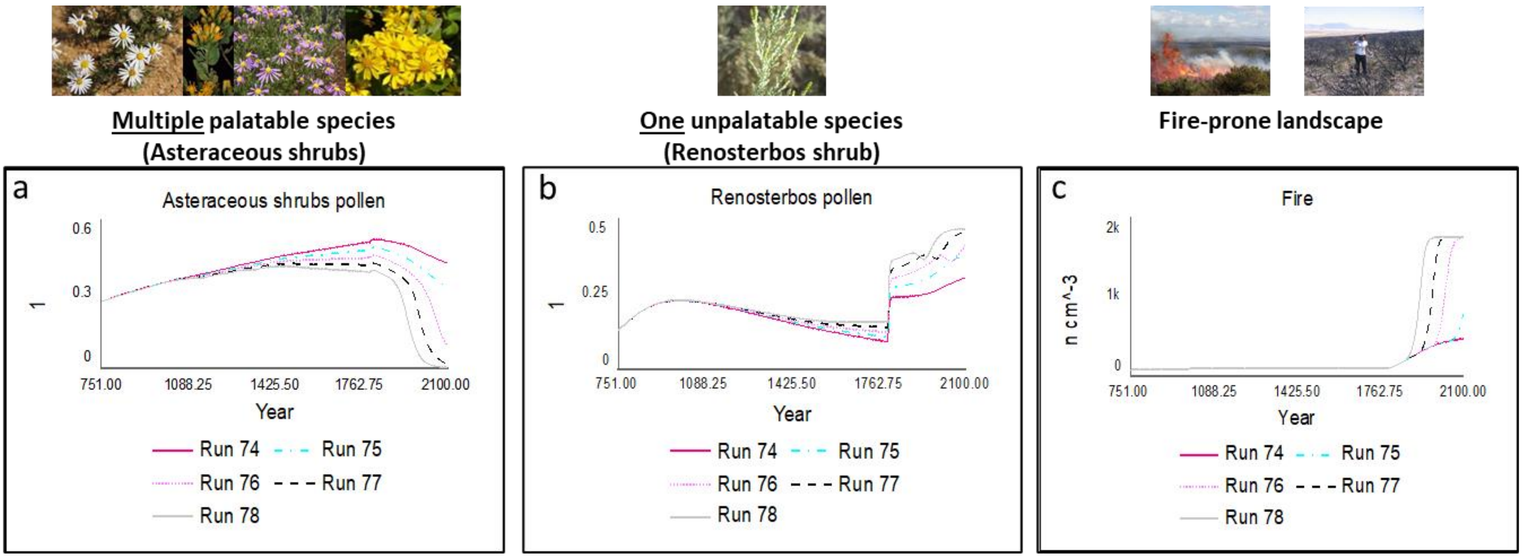


*Figure 10.25: Graphs showing an example of the behaviour sensitivity analysis outputs of the Ecological Model.*

*Table 10.10: Summary of the behaviour sensitivity analysis that was conducted on 18 parameters the Ecological Model.*

| | Exogenous variable tested for sensitivity | Current value (from Stella optimization) | Less 50% (current /2) | Plus 50% (current + less 50%) |
|---|---|---|---|---|
| Highy sensitivity | Asteraceous pollen relative decrease rate | 0.0159 | 0.0080 | 0.0239 |
| | Post-1800s fire relative increase rate | 0.2292 | 0.1146 | 0.3438 |
| | Post-1800s fire relative decrease rate | 0.0517 | 0.0258 | 0.0775 |
| | Standard effect of grazing | 2.6455 | 1.3228 | 3.9683 |
| | Standard effect of fire on A | 1.0249 | 0.5125 | 1.5374 |
| Medium sensitivity | Pre-1800s Asteraceous shrubs pollen relative increase rate | 0.0031 | 0.0015 | 0.0046 |
| | Pre-1800s Renostebos pollen relative increase rate | 0.0225 | 0.0112 | 0.0337 |
| | Post-1800s Renostebos pollen relative increase rate | 0.2047 | 0.1023 | 0.3070 |
| | Pre-1800s Renostebos pollen relative decrease rate | 0.2340 | 0.1170 | 0.3510 |
| | Pre-1800s fire relative increase rate | 0.3636 | 0.1818 | 0.5453 |
| | Pre-1800s fire relative decrease rate | 0.1009 | 0.0504 | 0.1513 |
| | Standard effect of fire on R | 0.9998 | 0.4999 | 1.4996 |
| | Max Renosterbos | 0.61 | 0.31 | 0.77 |
| Low sensitivity | Post-1800s Asteraceous shrubs pollen relative increase rate | 0.0352 | 0.0176 | 0.0528 |
| | Post-1800s Renostebos pollen relative decrease rate | 0.2407 | 0.1203 | 0.3610 |
| | Standard effect of homogenization on fire | 1.1432 | 0.5716 | 1.7147 |
| | Potential max fire | 2200 | 1100 | 3300 |
| | Max grazing | 0.86 | 0.43 | 1.00 |