ADVERSE IMPACTS OF AGRICULTURAL EXPANSION ON HYDROLOGICAL AND NUTRIENT DYNAMICS IN A RENOSTERVELD LANDSCAPE – CAN NATURAL VEGETATION OFFER SOLUTIONS?

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Dissertation presented for the degree of Doctor of Philosophy in the Faculty of Science at Stellenbosch University.

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March 2023

DECLARATION

By submitting this research dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

The nature and scope of my contribution to each chapter were as follows:

Chapter	Nature of contribution	Extent of contribution (%)
Chapter 1	This introductory chapter was written solely by me with editing and suggestions from my promotor and co-promotor.	Jan de Waal 95% Adriaan van Niekerk 3% Jodie Miller 2%
Chapter 2	This literature overview chapter was written solely by me with editing and suggestions from my promotor and co-promotor.	Jan de Waal 95% Adriaan van Niekerk 3% Jodie Miller 2%
Chapter 3	This chapter was written by me. Conceptualisation, data acquisition, analysis and interpretation were all conducted by me with feedback and input from both my promotor and co-promotor.	Jan de Waal 95% Adriaan van Niekerk 4% Jodie Miller 1%
Chapter 4	This chapter was submitted as a journal article to the *Hydrological Sciences Journal* and is under review. It was co-authored by my promotor and co-promotor as well as Dr Andrew Watson who assisted in setting up the hydrological model and provided input on the JAMS software environment. Conceptualisation, data acquisition, analysis, interpretation and write up were conducted by me. Manfred Fink wrote the code for the MUSLE component used in the hydrological model.	Jan de Waal 90% Adriaan van Niekerk 2% Jodie Miller 3% Andrew Watson 5%
Chapter 5	This chapter was submitted as a journal article to *Environmental Monitoring and Assessment* and it under review. It was co-authored by my promotor and co-promotor. Conceptualisation, data collection, data analysis and interpretation were done by me.	Jan de Waal 95% Adriaan van Niekerk 2% Jodie Miller 3%
Chapter 6	This chapter is formatted for submission as a journal article and was co-authored by my promotor, co-promotor and Dr Cathy Clarke. Conceptualisation, sample collection, laboratory analysis of N and P and interpretation was conducted by me. Dr Clarke assisted in providing advice on appropriate analysis techniques and conducted the soil classification with me. The chapter's results are augmented by results from an honours project conducted by Phillip Myburgh in 2021.	Jan de Waal 90% Adriaan van Niekerk 2% Jodie Miller 3% Cathy Clarke 5%
Chapter 7	This synthesis chapter was written by me with input and feedback from my promotor and co-promotor.	Jan de Waal 90% Adriaan van Niekerk 5% Jodie Miller 5%

The COVID-19 pandemic interrupted the conducting of the research for this dissertation. During 2020 the university and its laboratories were closed for several months so that no data could be collected or analysed during that time. Furthermore, I (the candidate) contracted COVID in June 2021 and was unable to collect data for a month, resulting in an unfortunate gap in the data.

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Date: March 2023

SUMMARY

Agricultural systems deliver a range of products to human society including, food, fuel, textiles and pharmaceuticals. However, the global expansion of agricultural activities has resulted in several negative outcomes such as biodiversity loss, increased carbon emissions, topsoil erosion and water pollution. Thus, the degradation of natural landscapes due to agricultural transformation has resulted in a loss of ecosystem services over time by increasing habitat loss, nutrient movement, sedimentation of rivers and pesticide poisoning in non-target species. One of the most impacted landscapes in terms of agricultural transformation in South Africa is renosterveld vegetation. Lowland renosterveld is a small-leaved, evergreen shrubland found on the shale-rich, fertile soils of the south-western Cape of South Africa where it forms part of the Fynbos biome, a species-rich floral kingdom. Renosterveld typically occurs on fine-grained, clay-rich soils as opposed to the sandy, nutrient-poor soils on which fynbos is located. Agricultural expansion has resulted in the destruction of the indigenous renosterveld vegetation which now exhibits a great degree of fragmentation. This dissertation documents an investigation of the impact of agricultural expansion on the hydrological, sediment and water quality dynamics in the Overberg renosterveld landscape in the Western Cape. An evaluation is reported of whether conservation of this threatened vegetation can allow for the delivery of ecosystem services in vegetation buffers in terms of phytoremediation of nutrient inputs from agricultural slopes. The impact of changing landuse on hydrological characteristics of the area at a landscape level is examined first, followed by a case study of the Bot River by implementing a fully differentiated hydrological model with a sediment delivery component. Results confirm that hydrology on a landscape level has been greatly impacted by changes in landuse, while modelled soil erosion from the Bot River catchment depicts an increase in soil erosion from 22 t/km^2/year under natural conditions to 490 t/km^2/year under 2018 landuse. A one-year monitoring programme of the river was undertaken to evaluate changing dissolved nutrient dynamics down the river's long profile through the use of ion-chromatography and stable isotope analysis. The results of this analysis indicate that nutrient loading in the river is linked to agricultural landuses and that $NO_{[x]}$-N levels in the river vary seasonally and periodically exceed water quality guidelines for aquatic ecosystems. Finally, an assessment was made of the potential for natural vegetation buffer strips to mitigate nutrient inputs from agricultural hillslopes. This was performed by an analysis of soil samples via inductively coupled plasma atomic emission spectroscopy (ICP-AES), laboratory-based testing for bio-available phosphorus, nitrate and ammonium as well as isotope ratio mass spectrometer (IRMS) testing of N and C isotopic composition in soils. Results show that N concentrations in cultivated field and renosterveld soils are impacted by fertilisation of agricultural lands. There is significantly ($p < 0.05$) more P in cultivated fields than in renosteveld soils, while renosterveld soils have a significantly ($p < 0.05$) higher C content than cultivated fields, thus acting as a valuable carbon sink. Renosterveld fragments are shown to remediate polluted agricultural runoff, and so provide a valuable ecosystem service in the landscape.

KEYWORDS

Ecosystem services, landuse change, agriculture, impacts, nutrient loading, phytoremediation

OPSOMMING

Landboustelsels lewer 'n reeks produkte aan die menslike samelewing, insluitend voedsel, brandstof, tekstiele en farmaseutiese produkte. Die wêreldwye uitbreiding van landbou-aktiwiteite het egter verskeie negatiewe uitkomste tot gevolg gehad, soos verlies aan biodiversiteit, verhoogde koolstofvrystellings, bogronderosie en waterbesoedeling. Die agteruitgang van natuurlike landskappe as gevolg van landboutransformasie het dus mettertyd 'n verlies aan ekosisteemdienste tot gevolg gehad deur toenemende habitatverlies, voedingstofbeweging, sedimentasie van riviere en plaagdodervergiftiging in nie-teikenspesies. Renosterveldplantegroei is een van die landskappe in Suid-Africa wat die meeste deur landboutransformasie geraak word. Laeland-renosterveld is 'n kleinblaar, immergroen struikveld wat op die skalieryke, vrugbare gronde van die Suidwes-Kaap van Suid-Afrika voorkom waar dit deel van die Fynbos-bioom vorm, 'n spesieryke blommeryk. Renosterveld kom tipies op fynkorrelige, kleiryke gronde voor in teenstelling met die sanderige, voedingsarm gronde waarop fynbos geleë is. Landbou-uitbreiding het tot die vernietiging van die inheemse renosterveldplantegroei gelei wat nou 'n groot mate van fragmentasie vertoon. Hierdie proefskrif dokumenteer 'n ondersoek na die impak van landbou-uitbreiding op die hidrologiese, sediment- en waterkwaliteitdinamika in die Overbergse renosterveldlandskap in die Wes-Kaap. 'n Evaluasie van die potensiaal van hierdie bedreigde plantegroei om ekosisteemdienste in terme van fitoremediëring van voedingstofinsette vanaf landbouhange te lewer, word gerapporteer. Die impak van veranderende grondgebruik op hidrologiese kenmerke van die gebied word eers op 'n landskapvlak ondersoek, gevolg deur 'n gevallestudie van die Botrivier deur die implementering van 'n volledig gedifferensieerde hidrologiese model met 'n sediment-leweringskomponent. Resultate bevestig dat hidrologie op 'n landskapvlak grootliks deur veranderinge in grondgebruik beïnvloed is, terwyl gemodelleerde gronderosie vanaf die Botrivier-opvanggebied 'n gronderosie toename van 22 t/km^2/jaar onder natuurlike toestande tot 490 t/km^2/jaar onder 2018 grondgebruik uitbeeld. 'n Eenjarige moniteringsprogram van die rivier is onderneem om veranderende voedingstofdinamika langs die rivier se langprofiel te evalueer deur van ioonchromatografie en stabiele isotoopanalise gebruik te maak. Die resultate van hierdie ontleding bewys dat nutriëntlading in die rivier aan landbougrondgebruike gekoppel is en dat $NO_{[x]}$-N-vlakke in die rivier seisoenaal verskil en periodiek waterkwaliteitriglyne vir akwatiese ekosisteme oorskry. Laastens is 'n assessering gemaak van die potensiaal vir natuurlike plantegroeibufferstroke om voedingstofinsette vanaf landbouheuwels te versag. Dit is uitgevoer deur grondmonsters deur middel van induktief gekoppelde plasma atoomemissiespektroskopie (ICP-AES), laboratorium-gebaseerde toetsing vir bio-beskikbare fosfor, nitraat en ammonium asook isotoopverhouding massaspektrometer (IRMS) toetsing van N en C isotopiese samestelling in gronde, te ontleed. Resultate toon dat N-konsentrasies in bewerkte veld- en renosterveldgronde deur bemesting van landbougrond beïnvloed word. Daar is beduidend ($p < 0.05$) meer P in bewerkte landerye as in renosterveldgronde, terwyl renosterveldgronde 'n beduidende ($p < 0.05$) hoër C-inhoud as bewerkte landerye het, wat dus as 'n waardevolle koolstofsink dien. Renosterveldfragmente remedieer besoedelde landbouafloop en bied dus 'n waardevolle ekosisteemdiens in die landskap.

TREFWOORDE

Ekosisteemdienste, grondgebruikverandering, landbou, impakte, voedingstoflading, fitoremediëring

ACKNOWLEDGEMENTS

I sincerely thank:

- My promotors Professor Adriaan van Niekerk and Professor Jodie Miller for their constant support, guidance and input;
- Dr Andrew Watson for his encouragement and assistance throughout the research process;
- Professor Michael Cherry for his unwavering support and wisdom;
- Dr Cathy Clarke for her kindness, advice and willingness to help;
- The laboratory assistants at the Biogeochemistry Research Infrastructure Platform (BIOGRIP) facility and Department of Soil Science at Stellenbosch University – Janine Colling, Jared van Rooyen, Volante Moonsamy, Leana de Witt and Faiek Arnolds;
- Dr Pieter de Necker for his language editing services;
- Manfred Fink who wrote the code for the MUSLE component used in the JAMS/J2000 hydrological model;
- My colleagues in the Department of Geography and Environmental Studies for their reassurance and support; and
- My wonderful family, especially Debbi, Grace-Anne and Jan Ezra de Waal, for always being there for me and backing me to the hilt.

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ACRONYMS AND ABBREVIATIONS

AMD	acid mine drainage
ARC	Agricultural Research Council
AWC	available water-holding capacity
AWS	automated weather station
BIOGRIP	Biogeochemistry Research Infrastructure Platform
C&RT	classification and regression tree
CFR	Cape Floristic Region
C_V	coefficient of variation
DEM	digital elevation model
DFFE	Department of Forestry, Fisheries and Environment
DTC	demonstration test catchment
DWS	Department of Water and Sanitation
EC	electrical conductivity
EC_a	apparent electrical conductivity
EM	electromagnetic
EPIC	erosion productivity impact calculator
ET	evapotranspiration
GEMS/Water	global environment monitoring system for freshwater
GIS	geographical information systems
GMWL	global meteoric water line
HRU	hydrological response unit

HWSD	Harmonised World Soil Database
IAEA	International Atomic Energy Agency
IC	ion-chromatography
ICP-MS	inductively coupled plasma mass spectrometry
ICP/OES	inductively coupled plasma optical emission spectroscopy
IDW	inverse distance weighted
IHDM	Institute of Hydrology distributed model
InHM	integrated hydrologic model
IQR	inter-quartile range
IRMS	isotope ratio mass spectrometer
JAMS	Jena adaptable modelling system
KGE	Kling-Gupta efficiency
Km	Klapmuts
LAI	leaf area index
LGR	Los Gatos Research
LMWL	local meteoric water line
LPS	large pore storage
MDL	method detection limits
MPS	medium pore storage
MUSLE	modified universal soil loss equation
NCMP	national chemical monitoring programme

NEMP	national eutrophication monitoring programme
NMMP	national microbial monitoring programme
NPK	nitrogen, phosphorus, potassium
NSE	Nash-Sutcliffe efficiency
NSGA	non-dominated sorting genetic algorithm
ORCT	Overberg Renosterveld Conservation Trust
Pbias	per cent bias
PTFE	polytetrafluoroethylene
QC	quality control
REMP	river ecostatus monitoring programme
RHP	river health programme
RSD	relative standard deviation
RUSLE	revised universal soil loss equation
S	subsoil
SANLC	South African national land cover
SANS	South African national standards
SASS	South African scoring system
SAWS	South African Weather Service
SCS	Soil Conservation Service
SDG	sustainable development goal
SHE	Systéme Hydrologique Européen

SRP	soluble reactive phosphorus
SRTM	Shuttle Radar Topography Mission
Ss	Sterkspruit
SUDEM	Stellenbosch University digital elevation model
Sw	Swartland
SWAT	soil water assessment tool
T	topsoil
TCD	thermal conductivity detector
TLB	tractor-loaded backhoe
TMG	Table Mountain Group
Tu	Tukulu
TWQR	target water quality requirements
UK	United Kingdom
USLE	universal soil loss equation
WEPP	water erosion prediction project
WHC	water-holding capacity
WMA	water management area
XRF	X-ray fluorescence spectrometry

CHAPTER 1: INTRODUCTION TO THE STUDY

1.1 INTRODUCTION

Agricultural systems provide a range of products to human society including, food, fuel, textiles and pharmaceuticals (Power 2010; Zhang et al. 2007). Global population growth has resulted in the need for increased agricultural production and the consequent transformation of land to agricultural uses (Jellason et al. 2021; Laurance, Sayer & Cassman 2014; Xiang, Malik & Nielsen 2020). World food production has thus more than doubled since 1961 (Food and Agriculture Organization 2022; Tilman 1999) with agricultural landuse now accounting for approximately 33% of the earth's land surface (excluding Antarctica and Greenland) (Dudley & Alexander 2017). Tilman (1999) also reports that the global use of fertilisers has increased markedly between 1961 and 1997. Nitrogen application to agricultural soils increased by a multiple of 6.9, while phosphorus fertilisation increased 3.5-fold. Furthermore, population growth is expected to continue resulting in a global population increase from 7.7 billion in 2019 to 9.7 billion in 2050 (United Nations 2019; Xiang, Malik & Nielsen 2020), so placing further pressure on agricultural production. However, in pursuit of greater productivity, the global expansion of agricultural activities has resulted in several negative outcomes such as biodiversity loss, increased carbon emissions, topsoil erosion and water pollution (Dudley & Alexander 2017; Duru et al. 2015). These unfavourable outcomes have contributed to a loss of ecosystem services over time by increasing habitat loss, nutrient movement, sedimentation of rivers and pesticide poisoning in non-target species (Zhang et al. 2007).

In South Africa renosterveld vegetation is one of the most impacted landscapes regarding agricultural transformation (Curtis 2013; Ntshanga, Procheş & Slingsby 2021; Topp & Loos 2019). Renosterveld is a small-leaved, evergreen shrubland – part of the Fynbos biome that is a species-rich floral kingdom – that typically occurs on fine-grained, clay-rich soils as opposed to the sandy, nutrient-poor soils on which fynbos is typically located (Cowling 1983; Cowling & Hilton-Taylor 1994; Moll et al. 1984; Rebelo et al. 2006). In comparison to fynbos vegetation, renosterveld is dominated by asteraceous shrubs such as *Elytropappus rhinocerotis* (renosterbos), grasses and geophytes, while *proteaceae*, *ericaceae* and Cape restios are largely absent (Curtis 2013; Cowling 1983). Owing to their distribution on fertile soils, renosterveld landscapes have experienced high levels of agricultural transformation over the past 300 years with only 4-10% of their original spatial extent remaining, most of which is found on privately owned land (Curtis 2013). The development of mechanised, large-scale agriculture after World War II massively accelerated the conversion of pristine renosterveld to pasture, oilseed and cereal crop land uses. This significant agricultural transformation has caused the renosterveld biome to become severely fragmented and as such, renosterveld is restricted to areas either too steep or too rocky to plough (Figure 1.1) (Topp & Loos 2019).

Source: Author

Figure 1.1 A renosterveld fragment surrounded by canola fields, Bot River, South Africa.

Such fragmentation of biomes has noteworthy impacts. For instance, the numbers of wild animals residing in the fragments decreases as fragment area reduces, while increased distance between fragments results in a reduction in animal movements between these islands (Haddad et al. 2015). Some species may however benefit from a loss of competition or predation, as well as changes in disturbance regimes (Haddad et al. 2015; Haddad et al. 2014; Laurance et al. 2002). This is evident in the renosterveld where fragmentation has benefited some species while being detrimental to many others (Topp & Loos 2019). For example, the blue crane (*Anthropoides paradiseus*) has fared well thanks to the species' preference for artificial grassland habitat (Figure 1.2) (McCann, Theron & Morrison 2007). By comparison, pollinators have been adversely affected by fragmentation (Donaldson et al. 2002) and floral species richness has dropped alarmingly with increased fragmentation (Topp & Loos 2019). Justified concern has been raised about the impact of habitat loss on black harriers (*Circus maurus*) that are forced from lowland renosterveld through agricultural transformation (Curtis, Simmons & Jenkins 2004). Thus, among the other impacts of agricultural expansion into renosterveld vegetation, the loss of biodiversity is especially disquieting (Donaldson et al. 2002; Kemper, Cowling & Richardson 1999; Winter, Prozesky & Esler 2007).

Source: Author

Figure 1.2 Blue cranes – a common sight in renosterveld landscapes.

Further environmental impacts relate to the loss of valuable ecosystem services offered by indigenous vegetation, including a loss in carbon sequestration (Mills et al. 2013) and the lowered infiltration of rainfall into soils (O'Farrell, Donaldson & Hoffman 2009). When comparing total carbon in fallow fields, active fields and renosterveld vegetation fragments, Mills et al. (2013) found that active (farmed) fields store less carbon that renosterveld fragments (69 Mg C/ha vs 84 Mg C/ha). Renosterveld convincingly offers a valuable ecosystem service regarding carbon sequestration. Renosterveld fragments also demonstrably reduce surface runoff, increase infiltration and assist in the retention of topsoil, thus providing other valuable ecosystem services (O'Farrell, Donaldson & Hoffman 2009). Moreover, intensive cropping in renosterveld landscapes has resulted in the salinisation of soils and water (De Clercq, Fey & Jovanovic 2009) through salt decantation processes similar to those documented in Australia (Bugan, De Clercq & Jovanovic 2010; De Clercq, Fey & Jovanovic 2009; Hingston & Gailitis 1976).

Landuse changes in renosterveld landscapes such as the Overberg, Western Cape[1], have also impacted adversely soil chemistry, structure and composition and comparable detrimental impacts on the quality and quantity of water moving through the Overberg landscape via rivers and other watercourses are very likely to be occurring. The conservation of watercourses has been identified as a priority in the renosterveld landscape because of their role as crucial ecological corridors (Curtis 2016; Overberg Renosterveld Conservation Trust 2020). These seasonally-wet areas, streams and rivers act as valuable ecological corridors that connect isolated fragments or islands of renosterveld vegetation by virtue of the relatively low levels of disturbance in these habitats as many have never been ploughed. Most of the Overberg Rûens Renosterveld and the watercourses are located on privately owned agricultural land and it is imperative that landowners are motivated to protect these systems (Topp & Loos 2019; Winter, Prozesky & Esler 2007). A mechanism to encourage the conservation of renosterveld adjacent to watercourses on private lands is to establish whether natural vegetation units can offer additional ecosystem services, such as the remediation of degraded water quality (O'Farrell, Donaldson & Hoffman 2009).

There is a paucity of literature on the biogeochemical–landscape–hydrological relationships or potential ecological services provided by the biome (Egoh et al. 2008; Le Maitre et al. 2007; Reyers et al. 2009). A crucial gap is the lack of understanding of the impacts the transitioning from natural vegetation to cultivated fields have on hydrological functioning and nutrient dynamics. Moreover, little is known about the ecological services regarding water purification rendered by pristine renosterveld (Jacklin, Brink & De Waal 2020). It is thus incumbent on researchers to determine the effects pristine renosterveld acting as vegetation buffers between cultivated hillslopes and waterbodies have on water purification, desalinization, surface flow rates, sediment runoff and hydrological connectivity of slope elements in the hydrological context of the Overberg region.

Four possible avenues of research in the watercourses in the Overberg Rûens are suggested by the foregoing discussion. They are:

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[1] The Overberg region is bordered by the Hottentots Holland Mountains in the west, Riviersonderend Mountains in the north, the Breede River mouth to the east and the Atlantic and Indian Oceans in the south. Indigenous vegetation in the region is dominated by lowland (Rûens Shale) renosterveld fynbos. The region is depicted in Figure 1.3.

(1) The hydrological component of watercourses and their relationship to landuse changes in the Overberg;

(2) The occurrence of soil erosion on the surrounding hillslopes that leads to increased turbidity in watercourses;

(3) The potential of eutrophication[2] in some areas due to high levels of agro-chemical runoff (fertilisers) given the evidence of indicators of high nutrient loads in Overberg watercourses, such as the prevalence of plant species like kikuyu grass and *Typha* species, as well as alien herbaceous species, grasses and forbs owing to their affinity for high nutrient loads; and

(4) The prospects for renosterveld buffers to mitigate the various harmful impacts on the local watercourses warrants investigation.

Furthermore, degraded watercourses have been identified by the Overberg Renosterveld Conservation Trust (Curtis 2016; Overberg Renosterveld Conservation Trust 2020) as having more potential for restoration than the disturbed renosterveld thanks to the presence of seedbanks and geophyte vegetation. Unfortunately, the deposition of sediment and nutrient loads into these watercourses has not been quantified and, moreover, the potential use of the renosterveld vegetation adjacent to these rivers to control erosion, limit salinisation and maintain water quality warrants investigation. These lacunae provide the rationale for this study.

1.2 STUDY RATIONALE

The transformation of renosterveld vegetation to agricultural land has adversely affected the biodiversity of the landscape (Topp & Loos 2019), the local soil and water chemistry (De Clercq, Fey & Jovanovic 2009) and the nutrient and soil dynamics in the region. There is thus a justifiable need to quantify and understand these negative consequences in order to enable and promote effective management. A constructive mechanism for managing these consequences is phytoremediation of watercourses using vegetation buffers (Cole, Stockan & Helliwell 2020; Liu, Zhang & Zhang 2008; Zhang et al. 2010). Many studies have shown the efficacy of vegetative buffers as active phytoremediatory zones between pollution sources and river systems. Buffers can effect decreases in nitrate concentrations in surficial (near-surface)

[2] "The artificial or natural enrichment of a river, dam or lake by an excessive influx of nutrients normally required for the growth of aquatic plants (such as algae)" (Van der Walt & Van Rooyen 1995, p.68).

groundwater by an order of magnitude or greater (Schnoor et al. 1995). While exploratory, laboratory-based experiments using some plant species present in the renosterveld landscape have revealed promising results regarding phytoremediatory potential (Jacklin, Brink & De Waal 2020), the field-based efficacy of renosterveld buffer strips calls for further exploration.

The landscape of the Overberg region has undergone substantial agricultural transformation that has resulted in the extreme degradation of its watercourses. Consequently, a case study of a watercourse, the Bot River, in the Overberg is envisaged to quantify the degree of hydrological change, erosion and nutrient loading that is attributable to this agricultural transformation and to investigate whether renosterveld buffers in the Overberg can aid the remediation and conservation of renosterveld. A successful demonstration of remediation and management in the Bot River can serve as a model for salvaging other degraded watercourses, not just in the Overberg but elsewhere in South Africa.

This study fits into the fields of global ecological services, drylands and land degradation. Global land cover alteration has been identified as a significant driver of ecosystem change and a limiting factor to idealised ecological functioning and service provision (Guerra, Rosa & Pereira 2019; Lambin, Geist & Rindfuss 2008; Reyers et al. 2009). Despite this global focus, very little research has identified the consequences of these changes on ecosystem services at local scales (Le Maitre et al. 2007; Reyers et al. 2009), most studies being typically descriptive in nature and limited to noting qualitative declines in ecosystem functioning (Reyers et al. 2009). Thus, an empirical evaluation and quantification of the impacts of landuse change on the hydrology, nutrient and sediment dynamics, as well as potential ecosystem services in the Overberg, can contribute to the strategic conservation objectives in the region. The appropriateness of the Bot River as a candidate for case study is assessed next.

1.3 THE BOT RIVER AS VENUE FOR A CASE STUDY OF RENOSTERVELD IN THE OVERBERG

A full evaluation of the effects of landuse change on the hydrological and nutrient dynamics of the Overberg region is severely constrained by the vast size of the area (the combined area of the Western, Central and Eastern Rûens Shale Renosterveld regions is 5970 km^2). For this reason a case study approach focusing on the Bot River was adopted as an example of how landuse change has affected the Overberg region. The Bot River was specifically selected because of the impressive mixture of various agricultural landuses and types of natural vegetation in the river’s drainage basin. This allowed for the determination of the sources of nutrients and their link to landuse. Furthermore, the river catchment is one of the few in the

Overberg renosterveld landscape that features a Department of Water and Sanitation (DWS) streamflow gauge. Little research has been conducted in the catchment and its proximity to the analytical laboratories at Stellenbosch University held promise to expedite the analysis of water samples.

Most of the extant research results relating to the Bot River focus on the Bot River vlei (lagoon) that is an important habitat for a variety of species (Koop, Bally & McQuaid 1983). In recognition of the importance of the lagoon, detailed studies on the Bot River estuary were conducted in the early 1980s as part of a larger project (funded by the South African National Committee for Oceanographic Research) to assess the status and conditions of South African estuaries (Bally 1987; Koop, Bally & McQuaid 1983). The six-year research programme resulted in the drafting of a management plan for the estuary (Sloan, Branch & Bally 1985). The findings of the research were published in a special issue of the Royal Society of South Africa's journal in 1985 (Branch 1985).

The avenues the investigations followed in the 1980s included assessments of the physical and biological characteristics of the estuary. Specific studies on the physical dynamics of the estuary covered the geomorphology (Rogers 1985), bathymetry and sediments (Willis 1985), wind-driven circulation of the lagoon (Van Foreest 1985), estuarine processes (Van Heerden 1985) and the hydrology of the system (Fromme 1985). Analyses of the biological dynamics of the estuary concentrated on the primary productivity of the estuary (Bally, McQuaid & Pierce 1985), zooplankton (Coetzee 1985), fish fauna (Bennett et al. 1985; Bennett 1985), bacteria (Roberts, Branch & Robb 1985), benthic communities (De Decker 1987) and the use of the estuary by waterbirds (Heÿl & Currie 1985). Special attention was given to the impact of the artificial opening of the estuary mouth on the biological dynamics and management of the system (Branch et al. 1985). Examination of nutrient levels in the estuary (Koop, Bally & McQuaid 1983) revealed that nutrient levels in the Bot River lagoon were relatively low compared to those of the nearby Palmiet River and the Berg River estuary. Notably, while NO_3^-, NH_4^+ and PO_4^{3-} were all present in the lagoon, there was no clear evidence of stratification of nutrients within the water column (Koop, Bally & McQuaid 1983).

A detailed description of the physical and chemical characteristics of the estuary given by Bally & McQuaid (1985) noted considerable fluctuations in the concentration of nitrates in the estuary as a response to rainfall, with sustained highs during the winter months. They also observed that the rivers in the Bot River catchment had low nutrient and salt contents. More recent research on the hydrodynamics of the Bot River (Van Niekerk, Van der Merwe & Huizinga 2005) considered the bathymetry of the estuary and provided an analysis of its hydrodynamics.

The Bot River vlei has also featured in several regional studies on South African estuaries (Hoveka et al. 2016; Montoya-Maya & Strydom 2009; Whitfield 1990; 1992).

In contrast to this extensive body of research results on the estuary there is limited information available on the river upstream of the estuary and about the larger catchment. There is only sparse nutrient data available for the river, despite the DWS having three biomonitoring sites in the catchment (Herdien et al. 2006), as well as historical water quality data made available on the DWS data portal (Department of Water and Sanitation n.d.) for the Roode Heuwel site (streamflow gauge G4H014). The impact of landuse change on water quality in the Bot River has not been fully quantified, especially when considering changes in nutrient levels down the river's long profile, and no fully distributed hydrological-erosion model has been developed for the catchment. Against this background, the next section is given to formulating the research problem and its associated questions.

1.4 RESEARCH PROBLEM

Agricultural practices evidently affect hydrological and erosion responses, as well as the nutrient fluxes in rivers (Arellano-Aguilar et al. 2017; Griffin 2017). Given that the impacts of agricultural expansion on the rivers in the Overberg region are poorly understood and remain wholly unquantified, an investigation into the effect of agricultural expansion on hydrology and nutrient levels in the Overberg is indeed warranted. Furthermore, if the application of fertilisers to fields is a factor driving the degradation of water quality in the Overberg landscape, methods to mitigate such impacts urgently call for exploration. In the renosterveld landscape transformation has led to fragmentation and most fragments are adjacent to watercourses. Hence they can be considered as vegetation buffers. It is reasonable to hypothesise that renosterveld fragments (these buffers) offer a valuable ecosystem service by reducing the concentrations and impacts of agricultural pollution on the natural system through phytoremediation (Jacklin, Brink & De Waal 2020). The results of an evaluation of the potential for renosterveld fragments to act effectively as buffers that remediate water pollution will contribute positively to motivating the need for conservation of these fragments. The relevant research questions are:

(1) How has landuse change affected catchment hydrology across the renosterveld landscape?
(2) How does the presence of cultivated fields affect soil erosion and sediment delivery to river systems in the Overberg?

(3) Are the spatio-temporal impacts of fertiliser application to cultivated fields on the freshwater systems in the Overberg quantifiable?
(4) What are the viable options for remediating water quality issues in the Overberg landscape?

These specific research questions should address some of the research gaps that exist in the literature concerning the impacts of agricultural practices on the biogeochemical status and hydrological functioning of the Overberg landscape. Furthermore, the research questions inform the aim and objectives set for this study.

1.5 AIM AND OBJECTIVES

The aim of this dissertation is to determine the impacts of agricultural expansion on hydrological, sediment and nutrient dynamics in a renosterveld landscape and to evaluate whether pristine renosterveld vegetation downslope of cultivated fields offers any ecosystem services to mitigate these impacts. This aim will be achieved by pursuing four research objectives, namely:

(1) Determine the impact of agricultural expansion on the landscape level hydrological functioning of rivers and catchments in the Overberg renosterveld landscape relative to natural reference conditions of native vegetation.
(2) Follow a case study approach to investigate the impact of climatic variability and landuse change on hydrological and sediment dynamics in a selected renosterveld catchment, the Bot River.
(3) Quantify the level of nutrient loading in the Bot River and evaluate the spatio-temporal variations thereof to determine the impact of agriculture on water quality in the Overberg.
(4) Examine whether strips of renosterveld vegetation located between cultivated fields and waterbodies are able to reduce sediment and nutrient delivery to the waterbodies in the Bot River catchment.

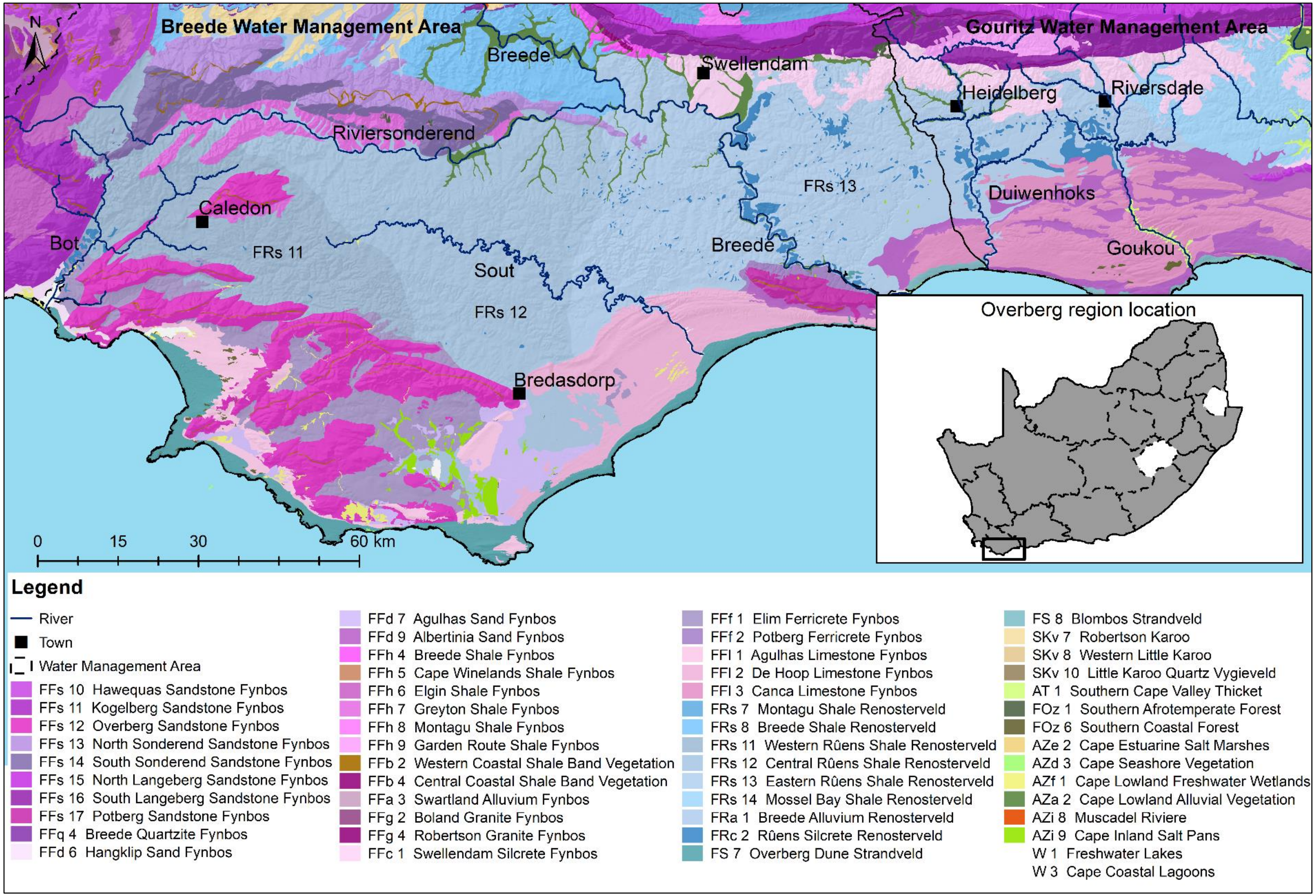
The Bot River catchment serves as an illustrative example of a transformed catchment in the Overberg region. It is anticipated that agricultural impacts in the Bot River are indicative of those experienced elsewhere in the Overberg region.

1.6 STUDY AREA DESCRIPTION

The Overberg in the Western Cape is home to a unique landscape of endangered Rûens Shale Renosterveld vegetation, extensive dryland agriculture, particularly oilseed and cereal crops, all overlying distinctive lithology (Figure 1.3).

Overberg Rûens Shale Renosterveld[3] is divided up into three predominant floristic (vegetation) areas – Western (FRs 11), Central (FRs 12) and Eastern (FRs 13) Rûens Shale Renosterveld (Figure 1.3) (Rebelo et al. 2006). Although this research focuses on the Rûens Shale Renosterveld landscape (stretching between the Bot River-Caledon area in the west to Bredasdorp in the south and Riversdale in the east), a comprehensive evaluation of the impacts of agriculture on all the area's watercourses was unfeasible given time and monetary constraints. Consequently, a case study approach was adopted focusing on the smaller Bot River catchment (865 km^2) located in the Western Rûens Shale Renosterveld vegetation unit (Figure 1.3). The results of this case study are deemed transferable to the other river catchments in the region owing to the similarity of the environmental contexts. Within the Overberg renosterveld landscape the watercourses and seasonally-wet areas are dominated by different (azonal) types of vegetation (as described by dominant species). Four types are noteworthy, namely (i) *juncus* reed beds – beds of rushes that closely resemble grasses or sedges; (ii) *phragmites* reed beds – large perennial grasses (reeds) located in wetland areas; (iii) *sarcocornia* plains – small flowering shrub-type vegetation; and (iv) thicket – dense stands of tall shrubs and trees. Because the soils of the Overberg region are nutrient-rich, the area is dominated by wheat and livestock agriculture (Herdien et al. 2006). This has led to substantial degrees of degradation of the renosterveld vegetation in the area.

[3] Renosterveld accounts for 29% of the spatial extent of the Fynbos Biome and most renosterveld vegetation units (86%) are found on top of an underlying shale geology – hence shale renosterveld. Renosterveld can also be located on other substrates such as granite and dolerite, alluvium and silcrete and limestone derived soils (Rebelo et al. 2006). Rûens Shale Renosterveld refers to shale renosterveld vegetation units located on the Rûens Shale geological formation (part of the Bokkeveld Group) located in the Overberg. The Overberg is also home to other renosterveld vegetation units as depicted in Figure 1.3.



Source: Mucina, Rutherford & Powrie (2005)

Figure 1.3 Regional study area depicting vegetation units found in the Overberg area.

This section overviews the characteristics of the Overberg landscape, while the specific site conditions are described in Chapters 4, 5 and 6. First, the geology, soils and topography of the Overberg region is described (Section 1.6.1) followed by an overview of the region's hydrology (Section 1.6.2), vegetation and climate (Section 1.6.3) and landuse (Section 1.6.4). Last specific attention is given to the Bot River catchment – the site of the case study conducted for this dissertation.

1.6.1 Geology, soils and topography

Most of the Overberg region is underlain by a bedrock of Bokkeveld Group shales (Rebelo et al. 2006) divided into the Western, Central and Eastern Rûens Shale formations. The Rûens region has an undulating, hilly topography in the west and a steeper, more dissected topography in the east. Altitude fluctuates between 200 and 300 m (Rebelo et al. 2006). In areas of higher elevation, silcrete and ferricrete form hard, flat-topped outcrops in the Rûens Shale landscape. The region is characterised by the presence of lithic, duplex, plinthic, gleyic and cumulic soils within the various land types (Fey 2010). Shallow, stony lithic soils dominate the region, typically forming the convex slope crests and gentler footslopes commonly found in the area (Fey 2010). Plinthic (cemented, iron oxide rich), gleyic (anaerobic, wetland) and cumulic (immature) soils also exist in the landscape. In the Western and Central Rûens Shale Renosterveld, clays and loams produced from the Bokkeveld Group shales are common, particularly the Glenrosa and Mispah forms (Fey 2010; Rebelo et al. 2006). The Eastern Rûens are characterised by clays and loams derived from Bokkeveld shales and Uitenhage Group clastics (Rebelo et al. 2006). The presence of duplex soils (also found in the landscape) indicates the enrichment of the subsoil with clay. There is thus a marked increase in clay composition in the B horizon compared to the horizon above (Fey 2010), often making the subsoil a limiting factor to both root growth and the movement of water through the soil. Consequently, the overland flow contributions to streamflow may be far greater than the infiltration and throughflow of water in the soil although shallow throughflow may be a substantial drainage component in soils overlaying shale because of the low permeability of the subsoil.

1.6.2 Hydrology

Much of the Overberg falls within the Breede-Gouritz Water Management Area. Two major rivers in the region are the Breede and Riviersonderend, although much of the Overberg is characterised by non-perennial streams with low flow rates and seasonally-wet areas (Herdien et al. 2006; River Health Programme 2011). The Lower Breede system is a meandering, lowland environment in the Eastern Rûens Shale Renosterveld which is joined by smaller, eastward-flowing, saline rivers stemming from the western and eastern Overberg (River Health Programme 2011). The Western Rûens Shale Renosterveld area features shorter rivers flowing coastwards through cereal croplands in the interior, whereas the eastern region contains more turbid, saline rivers (River Health Programme 2011). Rivers in the eastern Overberg contribute to a large number of wetlands and endorheic[4] systems in the coastal belt, such as the De Hoop vlei, where the flat topography makes the delineation of catchment boundaries a challenging exercise. Erosion rates in the Overberg renosterveld vegetation units are thought to be relatively low (Rebelo et al. 2006) with estimates of 16-450 t/km^2/year (Msadala et al. 2010) and 150 t/km^2/year (Rooseboom 1975). However, nationally soil erosion is increasing as a result of changes in landuse and the degradation of natural vegetation systems (Msadala et al. 2010).

1.6.3 Vegetation and climate

The distribution of vegetation usually shows marked similarities to that of soil so that soil type can be described to be a major determinant of the distribution of vegetation communities in the Overberg region (Thwaites & Cowling 1988). Shale renosterveld is the dominant veld type in the Overberg. Renosterveld is an evergreen shrubland or grassland, principally covered by asteraceous shrubs with a grass understorey and an abundance of geophytes (Boucher 1980; Moll et al. 1984). The floral diversity of renosterveld and number of geophytes increase with the frequency of veld fires and soil fertility (Kruger 1979). Renosterveld is mostly found on the lowland, shale-rich, fertile soils in the Overberg and is part of the Fynbos biome, but typically occurs on fine-grained, clay-rich soils as opposed to the sandy, nutrient-poor soils on which fynbos is located (Curtis 2013; Curtis & Bond 2013). *Elytropappus rhinocerotis* (renosterbos), grasses and a wide diversity of geophytes are common in renosterveld vegetation units (Moll et al. 1984) while proteaceae, ericaceae and Cape restios are largely absent (Cowling 1983; Curtis 2013). The renosterveld of the Overberg tends to contain fewer geophytes and more C_4 grasses. Canopy cover does grade into thicket type vegetation in the east where the dissected topography inhibits the spread of fire (Moll et al. 1984). The Overberg region has three dominant vegetative areas: Western Rûens, Central Rûens and Eastern Rûens Shale Renosterveld. Remarkably, renosterveld has not been subjected to detailed analysis and classification of the vegetation structure despite the structural diversity of this vegetation type

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[4] Endorheic systems have no river outflow to an outside body of water (e.g. a river or ocean) and water losses occur only by evaporation or ground seepage.

(Rebelo et al. 2006). The complex interactions of local landforms, geology, climate and soil types create unique environments for the diverse range of species. However, the individuality of these habitats results in high levels of ecological sensitivity with narrow distribution ranges for many of the species (Herdien et al. 2006; Mustart, Cowling & Albertyn 2003). Seasonally-wet areas, streams and rivers act as crucial ecological corridors for renosterveld vegetation owing to the relatively low levels of disturbance in these habitats.

Rainfall varies between 400-500 mm/a in the west to 350-400 mm/a in the central areas and 275-400 mm/a in the east (Bailey & Pitman 2015). The Western and Central Rûens are characterised by winter rainfall (May to August) and dry summers (December to March), while the Eastern Rûens displays a more even distribution with slightly lower precipitation in the summer months (Bailey & Pitman 2015; Rebelo et al. 2006). The temperature ranges for all three regions are quite similar – typically between 5°C (July average minimum) and 27°C (January average maximum) (Rebelo et al. 2006). As a result, drylands agriculture is the dominant landuse in the Overberg.

1.6.4 Landuse

Substantial transformation of the renosterveld landscape has occurred over the last 300 years. In the 18th century large game and Khoikhoi cattle herds were replaced by livestock farming (Rebelo et al. 2006) which was eventually transitioned to cultivated fields. Currently, agriculture makes up the bulk of landuse activities in the region. Cereals such as wheat and barley are grown, while canola (an oilseed crop) has been introduced more recently (Figure 1.4) (Leeuwner et al. 2003).

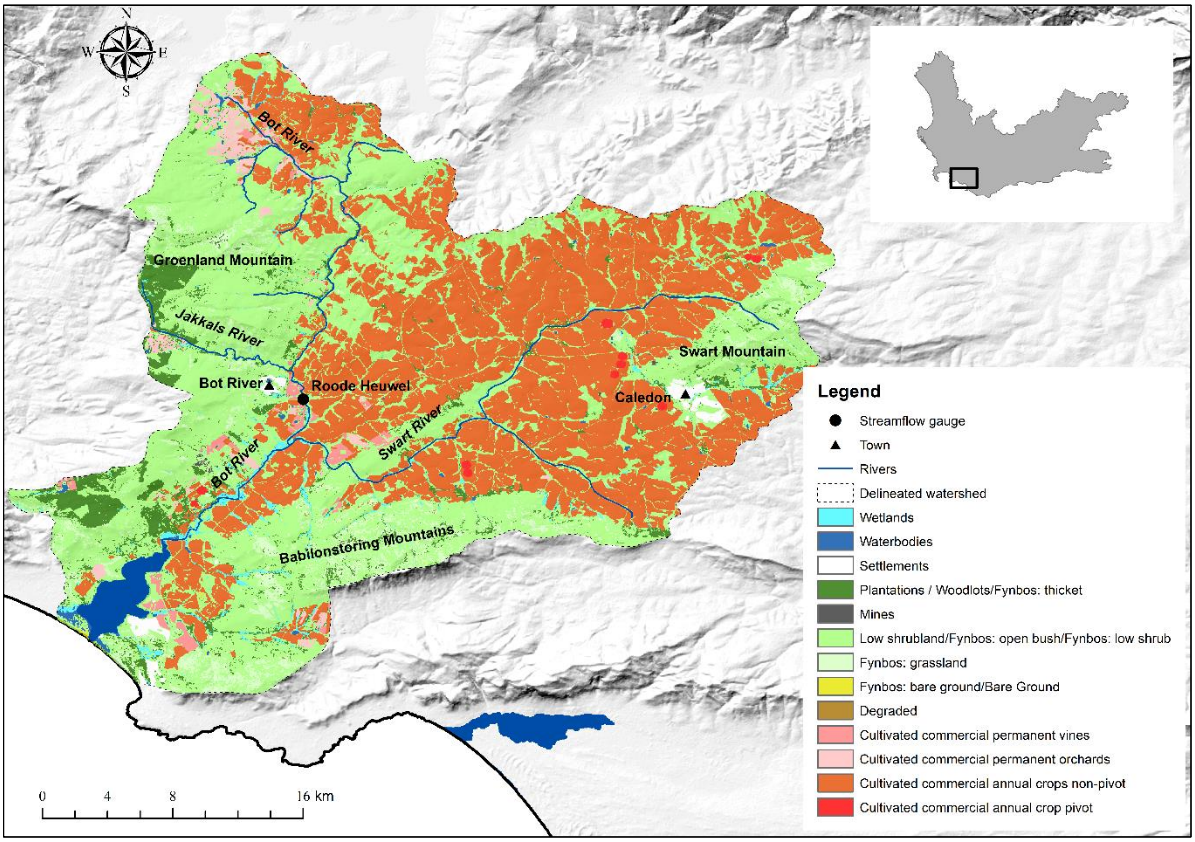
Source: Author

Figure 1.4 Fields of canola in the Bot River catchment.

Traditionally a wheatland–fallow system was practiced, however many farmers now alternate cereal crops with dryland pasture grazed by cattle and sheep (Herdien et al. 2006). Few (4-10%) pristine renosterveld habitats remain, most of which are small islands located on private property (Curtis 2013; Curtis 2016). These features constrain the determination of effective and appropriate conservation measures and management practices.

1.6.5 Focus on the Bot River

The Bot River (Figure 1.5) is a relatively small, semi-arid to dry subhumid catchment located in the Overberg. There are various explanations for the name 'Bot' and Burman (1970) has suggested that it derives from the early Cape butter ('botter' in Afrikaans) merchants who traded tobacco, arrack (an alcoholic spirit), coffee and glass beads for the butter produced by the indigenous inhabitants who often encamped in the area.



Data source: Department of Forestry, Fisheries and the Environment (2018)

Figure 1.5 The Bot River catchment in the Overberg, Western Cape.

The Bot River catchment has an area of about 900 km^2 (Bally 1987; Van Niekerk, Van der Merwe & Huizinga 2005). The river begins on flat topography after which it winds through undulating hills before exiting at the coast in a vlei near the town of Hawston. Streamflow is greatly augmented by the confluence of mountain streams from the western edge of the catchment (in the Groenland Mountain) with the main trunk stream. The major tributaries to the Bot River are the Swart and Jakkals Rivers flowing in from the east and west respectively (Figure 1.5). The Babilonstoring Mountains are the southern watershed divide. The average annual discharge, as measured by the DWS Roode Heuwel gauge (G4H014-A01) (Figure 1.6) near the town of Bot River, is 22.6 x 10^6 m^3/a (Department of Water and Sanitation 2021). The catchment is in a winter rainfall zone and the mean annual precipitation on the Groenland Mountain is 1100 mm/a (1000 m altitude) and 450 mm/a at lower elevations towards the centre and east of the catchment (0-200 m altitude) (Bailey & Pitman 2015). The Swart River to the east of the catchment is often dry in the summer months. The catchment's geology is dominated by the Western Rûens Shale formation in the low-lying areas and Table Mountain Group (TMG) sandstone at higher elevations.

Source: Author

Figure 1.6 DWS Roode Heuwel gauge (G4H014-A01) on the Bot River.

Western Rûens Shale Renosterveld is the principal natural vegetation unit in the area occurring on nutrient-rich, shale-derived soils, whereas the nutrient-poor slopes of the mountainous areas are largely covered by mountain fynbos (Curtis & Bond, 2013; Rebelo et al., 2006). Much of the land area once covered by renosterveld has been converted to cereal and oilseed agriculture (Curtis 2016; Curtis 2013). Common crops in the catchment are canola, wheat, lucerne and oats. The riparian zones of the Bot River are overrun by a multitude of invasive plant species, while extensive reed beds of *Phragmites* and *Scirpus* occur south of the confluence of the Bot and Swart Rivers where the catchment gradient lowers substantially (Koop, Bally & McQuaid 1983). Past chemical analyses were conducted on surface waters from the Jakkals River and water collected an unknown point along the Bot River (Bally & McQuaid 1985). The water of the former river was reported as being notably acidic. Mean NO_3^-, NH_4^+, PO_4^{3-} and SO_4^{2-} concentrations were reported as 0.03, 0.08, 0.02 and 11.46 $mg.l^{-1}$ for the Jakkals River and 0.27, 0.05, 0.02 and 26.67 $mg.l^{-1}$ for the Bot River (Bally & McQuaid 1985).

An overall description of the Overberg region and the Bot River catchment reveals the environmental setting in which this research is conducted. The overarching research methodology and design was partly informed by the types of soils, vegetation units and watercourses as well as data accessibility/availability in the region with a view to evaluating how agricultural landuses impact these features of the landscape.

1.7 RESEARCH METHODS AND DESIGN

The research adopted a mixed-methods approach that involved observation, testing, measurement, modelling and statistical analysis in determining the influence of agricultural expansion on hydrological, sediment and nutrient dynamics in the Overberg and defining whether naturally occurring renosterveld vegetation offers any ecosystem services to mitigate these impacts. The research was conducted in four phases, each with its own separate methods informed by relevant literature. The overall research design is illustrated in Figure 1.7. Each phase of the study is reported in a separate chapter.

Phase 1 investigated the impact of landuse change on hydrological conditions of small catchments across the Overberg by identifying physiographically similar catchments under two landuse scenarios (historical renosterveld vegetation and contemporary transformed landuse). Quinary[5] catchments were delineated and then grouped according to their similarity based on physiographic attributes through cluster analysis. Changes in cluster groupings among quinary catchments under renosterveld vegetation, when compared to clusters under contemporary landuse, were assumed to be indicative of the impacts of landuse change on these catchments. The specific methods used are described in Chapter 3.

Phase 2 involved a case study in which a fully distributed hydrological model of the Bot River was implemented. The hydrological model aimed to estimate various flow components

[5] The DWS has delineated primary, secondary, tertiary and quaternary catchments for South Africa. Quinary catchments (Maherry et al. 2013) are smaller catchments nested within quaternary drainage basins.

(overland flow, soil water and groundwater contributions to streamflow) and simulate runoff, streamflows and sediment dynamics in the Bot River catchment. The model was used to evaluate how changes in landuse over time influenced hydrological and sediment dynamics in the Bot River and to illustrate the impacts of agricultural transformation in the broader renosterveld landscape. A more detailed description of this case study's methods is given in Chapter 4. The chapter was written as an article for publication.

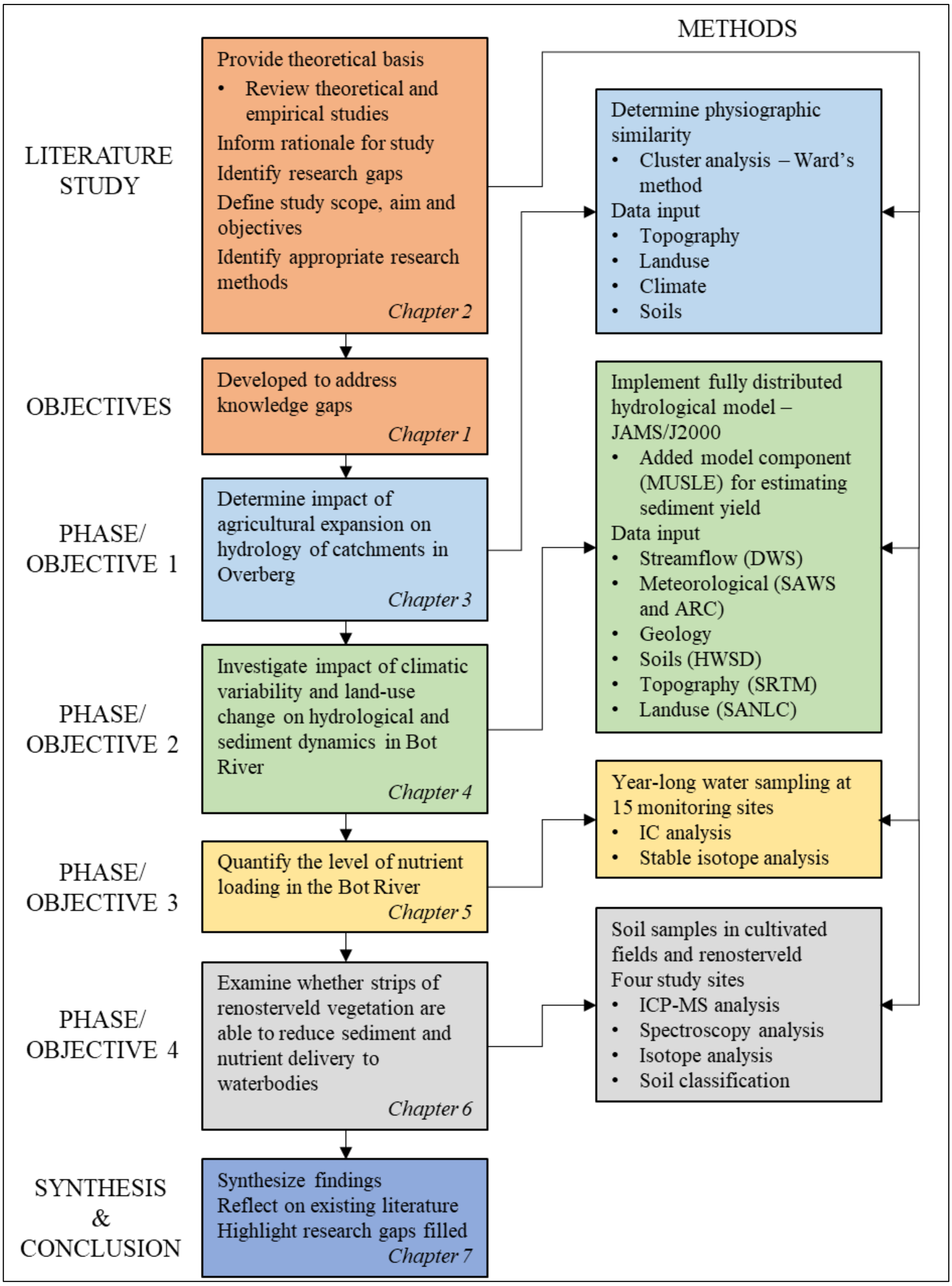


Figure 1.7 Research design for the study.

Phase 3 was another case study of the Bot River involving a one-year water-quality study of the river. Fifteen observation sites along the river's long profile were identified and water samples were collected during the dry summer and wet winter. Samples were tested for anions and cations as indicators of nutrient levels and fluxes in the river to assess the impacts of agriculture on river health. A full account of the methods applied is given in Chapter 5. The chapter was written as an article for publication.

Phase 4 aimed to evaluate the effect of renosterveld vegetation buffer strips between agricultural hillslopes and receiving waterbodies (rivers) on nutrient delivery to these watercourses. Four observation sites were selected in the Bot River area for the collection of soil samples in cultivated fields and natural renosterveld used to determine the sources of nitrogen in the landscape and changing nutrient levels down the catena[6]. The methods are described in Chapter 6. The chapter was written as an article for publication.

1.8 CONCLUSION

This chapter has introduced the renosterveld landscape and the challenges caused by historical agricultural transformation in the Overberg region. It established that although much research has been conducted on renosterveld vegetation, the focus has been on biodiversity and descriptions of the vegetation (Cowling 1983; Cowling & Hilton-Taylor 1994; Curtis & Bond 2013; Donaldson et al. 2002), the rehabilitation of renosterveld (Milton 2001) and the habitat and ecological functions of renosterveld (Mills et al. 2013; O'Farrell, Donaldson & Hoffman 2009). Research gaps in the understanding and quantifying of the impacts of agricultural transformation on nutrient and sediment dynamics in the Overberg have been identified. Furthermore, the unascertained role renosterveld buffers play in remediating the potential impacts of nutrients on local rivers is pointed out.

The filling of these knowledge gaps by evaluating the impacts of agricultural activities on catchment hydrology, sediment and nutrient dynamics in the renosterveld landscape using a case study of the Bot River constitutes the aim and objectives of this study. An assessment of the efficacy of renosterveld buffers in mitigating these impacts is also envisaged. The literature consulted in planning and undertaking the research is reviewed in the next chapter.

[6] A catena is a sequence of soils down a hillslope that have approximately the same age and are derived from similar parent material (Van der Walt & Van Rooyen 1995).

CHAPTER 2: OVERVIEW OF RELEVANT LITERATURE

2.1 INTRODUCTION

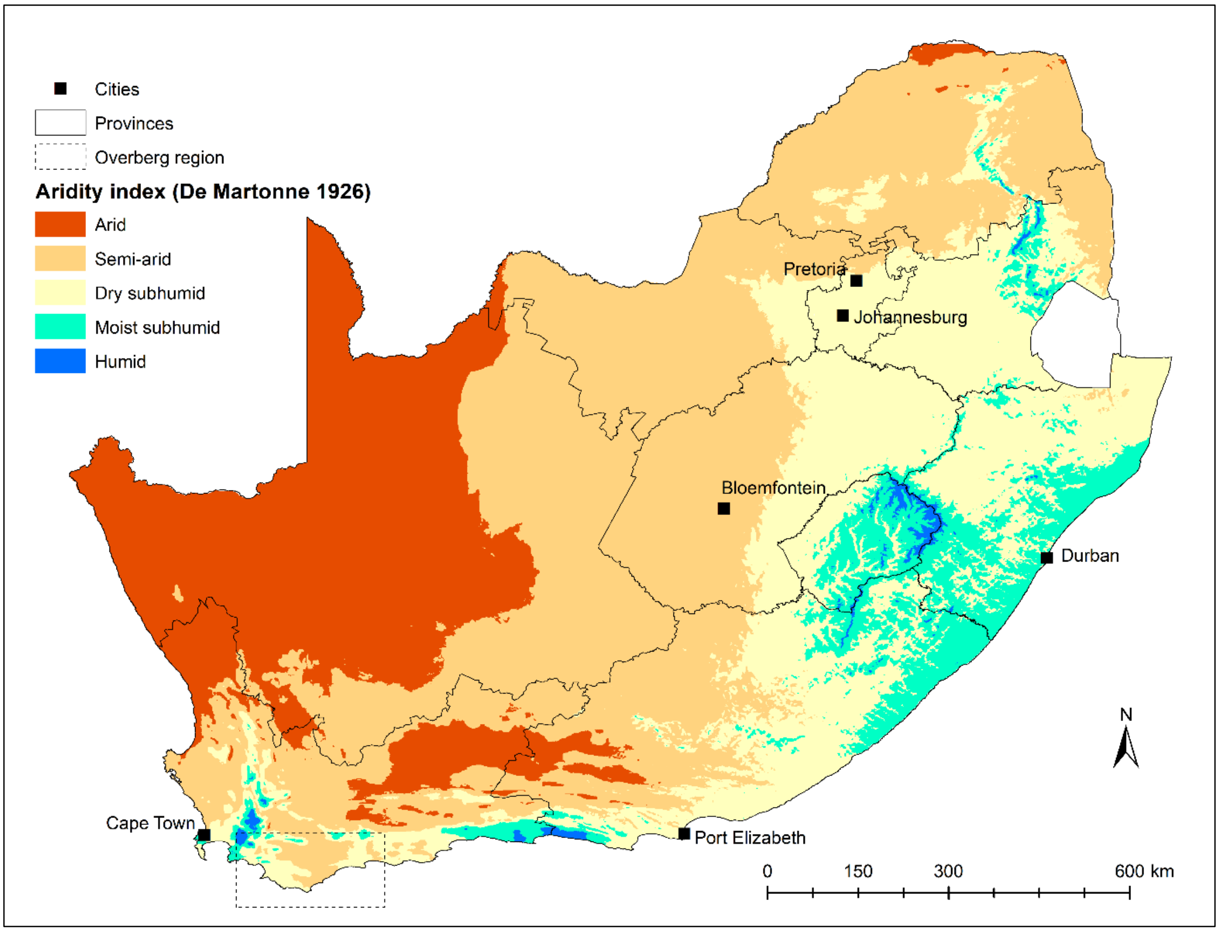
This chapter provides an overview of literature relevant to this research. First, the research is placed in the context of broader drylands ecosystems narrowing to focus on the renosterveld landscape as a dryland ecosystem (Section 2.2). Second, the fundamental hydrological processes that affect surface runoff and subsurface flow components and how these contribute to perceptual and conceptual hydrological and erosion models are discussed (Section 2.3). Third, the concept of ecosystem services is explored. The specific ecosystem service of interest to this research is the ability of renosterveld vegetation to remediate nutrient loads from agricultural hillslopes before they reach watercourses (Section 2.4). The fourth body of literature considered relates to phytoremediation (Section 2.5). Finally, a brief review of the monitoring and evaluation of water quality is considered (Section 2.6). Literature germane to the research objectives is explored further in Chapters 3, 4, 5 and 6.

2.2 DRYLAND ECOSYSTEMS

Drylands (arid ecosystems) extend over some 45% of the earth's land surface and constitute the world's largest ecosystem type (James et al. 2013; Schimel 2010). Drylands are classified as hyperarid, arid, semi-arid and dry subhumid depending on water availability (Prăvălie 2016; Safriel et al. 2005). Drylands consequently occur where annual potential evapotranspiration exceeds annual precipitation (Darkoh 2003; Hassan & Dregne 1997). They are described as critical ecosystems due to their tight water balance and limited water availability (Prăvălie 2016). Despite these ecosystems' vast extent and vulnerability to degradation (Dregne 2002), the literature on global change tends to concentrate on other ecosystems, such as the humid tropics where biomass and biodiversity are far greater and on the polar regions which are highly susceptible to a changing climate (Schimel 2010). By contrast, drylands receive less research attention because of their relatively low rates of biological activity and the limited biota. However, in some dryland ecosystems, such as in renosterveld vegetation (Cowling 1983; Rebelo et al. 2006), biodiversity can be strikingly high (Maestre, Salguero-Gómez & Quero 2012).

Dryland regions are susceptible to several environmental stressors (both natural and anthropogenic), including land degradation through erosion, vegetation and biodiversity loss through landuse transformations, increased salinisation and nutrient losses (Prăvălie 2016). African and Asian drylands are particularly at risk due to their vast spatial extent on these two continents and to the distinctive environmental disturbances they face (Prăvălie 2016). Despite their great extent and crucial importance, the effects of environmental changes on dryland systems remain under-researched (Maestre, Salguero-Gómez & Quero 2012).

South Africa is often reported as the 30th most dry country globally (De Kock 2021; Naidoo 2017; Republic of South Africa 2015), although this has been disputed by some scholars (e.g. Winter 2018). Large areas of the country are indeed classed as arid to semi-arid (Snyman 1998; Vetter 2009). A classification of the country's aridity is mapped in Figure 2.1, according to which the country is divided into five broad aridity classes – arid, semi-arid, dry subhumid, moist subhumid and humid. Notably, the Overberg region is characterised by semi-arid to dry subhumid conditions, firmly establishing the area as a dryland.



Data source: Council for Scientific and Industrial Research (2015)

Figure 2.1 Spatial distribution of aridity in South Africa according to the De Martonne (1926) aridity index.

Dryland vegetation is remarkably resilient to external stressors such as drought and fire and, has the innate capacity to recuperate rapidly from these events (Behnke, Scoones & Kerven 1993; Darkoh 2003). However, the pressures from agriculture through grazing and extensive cropping drastically reduce biodiversity, vegetation structure and resilience by the destruction and alteration of habitats (Adhikari et al. 2019; Darkoh 2003; 2018). The vegetation in southern

Africa is universally known for its abundance of species (Cowling 1983; Cowling et al. 1996; Gibbs Russell 1987; Thuiller et al. 2006), with 3.5% of the global floral diversity occurring in just eight plant diversity hotspots found in the region (Cowling & Hilton-Taylor 1994). The hotspot with the greatest species richness occurs in the Cape Floristic Region (CFR) located primarily in the Western Cape province of South Africa (Meadows 2000; Rebelo et al. 2006).

Despite its relatively small size, the CFR is one of most biodiverse habitats in the world and is characterised by substantial floral diversity (Cowling 1990). Approximately 9000 plant species are found in the CFR and many (70%) of these plants are endemic to the region (Giliomee 2006; Goldblatt 2000; Rebelo et al. 2006). In the South African context, the CFR covers only 4% of the country, yet constitutes some 44% of floral diversity in the region (Von Hase et al. 2003). Much of the CFR is classified as semi-arid, with subhumid areas confined to mountainous topography that generates orographic rainfall (Bennie & Hensley 2001). Low-lying areas in the CFR are typically classified as drylands.

The Fynbos biome draws its name from the dominant vegetation type – fynbos (Rebelo et al. 2006). It extends along the Cape Fold Mountains and occupies much of the coastal plain between mountains and adjacent oceans (Rebelo et al. 2006). The biome comprises three distinct vegetation types which are naturally fragmented by soil and climate distributions, namely fynbos, renosterveld and strandveld. Widespread agricultural development (especially on arable soils and slopes along the coastal plain) has led to a high degree of fragmentation in the region, particularly in renosterveld vegetation (Curtis & Bond 2013; Rebelo et al. 2006; Topp & Loos 2019).

Renosterveld landscapes account for 29% of the spatial extent of the Fynbos Biome (25% of the CFR) (Rebelo et al. 2006) and is described as an evergreen shrub or grassland. Small shrubs (such as the commonly found renosterbos – *Elytropappus rhinocerotis*) are underlain by a variety of grasses and bulbs (Boucher 1980; Curtis & Bond 2013; Moll et al. 1984). Renosterveld is characterised by the absence of proteas, ericas and restios which are more indicative of fynbos landscapes. As much as 86% of renosterveld vegetation is located on shale substrates with nutrient-rich, clayey soils (Curtis & Bond 2013; Kemper, Cowling & Richardson 1999; Rebelo et al. 2006). Lowland renosterveld has a characteristically homogenous grey appearance in the landscape due to the prevalence of low asteraceous shrubs. But despite its drab appearance renosterveld has one of the highest species diversities per unit of land area in the world (Curtis & Bond 2013), greater even than the neighbouring fynbos vegetation. Lowland renosterveld is typically grassier than fynbos and is well known for its geophytic diversity (Curtis & Bond 2013). The abundance and diversity of geophytes are hypothesised to be attributable to increasing soil fertility, dryness and fire frequency that lead to a greater diversity (Kruger 1979; Rebelo et al. 2006). Renosterveld habitats are delineated into two distinct geographical regions, namely West Coast Renosterveld and South Coast Renosterveld (Curtis & Bond 2013), both of which are located on shale-derived soils (Rebelo et al. 2006). The soils underlying South Coast Renosterveld are more fertile than those of West Coast Renosterveld, while eastern vegetation units in the South Coast Renosterveld receive more summer precipitation (Bailey & Pitman 2015) which results in a greater abundance of palatable (C_4) grasses. The Overberg region (which roughly covers lowland Rûens Shale Renosterveld in the South Coast region) receives a mean annual precipitation of 350-500 mm, and the Swartland region (where most of the West Coast Renosterveld is located) has a similar rainfall regime (Bailey & Pitman 2015), thus firmly establishing renosterveld landscapes as dryland ecosystems.

Whereas Curtis & Bond (2013) describe two distinct geographical areas for lowland renosterveld, Moll et al. (1984) earlier segregated renosterveld vegetation into four discrete biogeographical categories: (1) West Coast Centre Renosterveld, which is inclined to have less grass cover (mainly C_3[7] grasses) and a high diversity of deciduous geophytes; (2) South Coast Renosterveld, which often has fewer geophytes but more grasses (largely C_4); (3) Inland Mountain Centre Renosterveld, which is particularly arid with less cover than lowland types, with more succulents and C_4 grasses present in the landscape; and (4) Eastern Centre Renosterveld, which is relatively homogenous structurally and has a large component of C_4 grasses. Eastern Centre Renosterveld has many similarities with Albany Thicket and grassland vegetation types (Moll et al. 1984). This research focuses on the South Coast Renosterveld region and considers the ecosystem services that the remaining vegetation fragments in this ecosystem may have to offer. Of great concern is the impact of landuse change on the water quality and the hydrology of the Overberg landscape.

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[7] C_3 and C_4 grasses refer to the photosynthesis pathways these grasses use to produce their nourishment (the two most common being the C_3 and C_4 pathways) (Curtis & Bond 2013). Most shrubs in renosterveld exhibit C_3 photosynthesis as do grasses like perennial veldt grass and bokbaardgras. Grasses with a C_4 signature include red grass (rooigras) and dobo grass (buffelsrooigras) (Curtis & Bond 2013).

2.3 HYDROLOGICAL PROCESSES

This section is concerned with describing hydrological concepts germane to the research. First, an overview of hillslope hydrological processes is given (Section 2.3.1). Second, a discussion on the concept of hydrological connectivity is provided considering hillslope (Section 2.3.2) and subsurface flow (Section 2.3.3) connectivity. Fourth, a brief history of hydrological modelling in provided (Section 2.3.4). Finally, the hydrological characteristics of drylands are discussed (Section 2.3.5).

2.3.1 Hillslope hydrological processes

The classical model of hillslope hydrology outlined by Horton (1933) describes the soil surface as behaving as a sieve to divide rainfall into two components – overland flow (sometimes referred to as Hortonian overland flow and surface runoff which is driven by either saturation or infiltration excess) and groundwater flow (Chorley 1978). The conversion of rainfall to overland flow or groundwater flow is determined by the infiltration capacity of the soil and whether rainfall intensity exceeds this or not (Horton 1937; 1939; 1945). Horton contended that the maximum rate of rainfall infiltration into soils is a function of the permeability of the soil and the length of time rain has been falling (Chorley 1978; Horton 1933). Therefore, prolonged rainfall on slopes in a catchment will eventually produce overland flow after an initial abstraction to the soil. Hortonian overland flow is viewed as the major contributor to storm peaks in a river’s hydrograph and the primary driver of surface fluvial erosion (Chorley 1978; Ward 1967).

The nine principal components of the hillslope hydrological cycle are: (i) interception – the proportion of rainfall that does not reach the soil surface as it is intercepted by plants and other structures; (ii) evapotranspiration – water loss from transpiring plants, open water surfaces and soils; (iii) depression storage and detention – the volume of water that can be held in landscape depressions without running off; (iv) infiltration; (v) overland flow; (vi) soil moisture storage – defined as the amount of water that a field can retain in the soil in opposition to gravity; (vii) diffuse lateral soil water movement (soil throughflow) – the lateral movement of water within the soil pore spaces; (viii) concentrated lateral soil water movement – the movement of water in concentrated zones in the soil such as soil piping and fissures; and (ix) deep seepage – the recharge of groundwater aquifers through infiltration and vertical movement of water through soils (Chorley 1978). Rainfall will convert either to surface runoff or soil water as a function of precipitation depth-duration (Overeem, Buishand & Holleman 2008) and these hillslope properties. The partitioning of rainfall into surface and subsurface flow components is discussed in Section 2.3.1.1 and the consequent erosion of soils from hillslopes by surface runoff is described in Section 2.3.1.2.

2.3.1.1 Partitioning of rainfall into flow components

Because soil moisture is a major determinant of the partitioning of rainfall into Hortonian overland flow and infiltration components (Castillo, Castelli & Entekhabi 2015), a key factor influencing infiltration is the underlying soil in the catchment (Knapp 1978). Hillslope soils are complex and vary substantially down the catena. However, basic soil responses can be estimated according to soil characteristics. Soil moisture is stored in pore spaces in soils (Knapp 1978). Moisture is retained in the soil because the surface tension of the water found in these pores allows it to resist the influence of gravity and capillary action (although these forces are often too great to overcome) (Ward 1967). Surface tension force is related to the surface area of the water's meniscus within the pore space, thus the smaller the area, the longer the soil moisture remains in situ (Knapp 1978). Soil moisture can be measured directly or estimated based on several criteria, namely rainfall, solar radiation, environmental evaporative demand, slope topography, landuse and soil characteristics (Castillo, Castelli & Entekhabi 2015). This is common practice in modern hydrological models such as the soil and water assessment tool (SWAT) (Arnold et al. 1998) and the Jena adaptable modelling software (JAMS) (Krause 2001).

Most soil moisture is in a state of motion due to differences in input, storage and output (Knapp 1978). Regarding input, water will only infiltrate into the soil where a film of water is present at void entrances (Knapp 1978), meaning that when soils have been dried completely by evaporation they must be wetted before any meaningful infiltration can occur (with the exception of macrovoid spaces). As a result, soils with large, horizontally-aligned structural units (called peds) can often resist infiltration. This demonstrates that the antecedent soil moisture conditions are critical in determining the degree of infiltration that takes place (Knapp 1978). Where soil moisture is very high, soils are quickly saturated leading to the generation of overland flow. Conversely, where soil moisture is negligible, soils can often restrict infiltration, which also leads to surface runoff (Knapp 1978; Ward 1967). The upward movement of water in soils (capillary rise) (Wollny 1887) occurs as evaporation and transpiration generate a suction gradient at the surface (Ward 1967). As this force overcomes gravity and soil moisture surface tension, water rises through the soil towards the root zone and surface, thus creating a vertical movement of soil moisture. The amount of soil moisture lost is determined by the rate of drying at the surface (Penman 1941; 1948). If surface evaporation is relatively slow, capillary rise is able to remain in equilibrium with moisture lost at the surface. If, however, rapid drying takes place at the surface, capillary rise may be unable to keep up with surface losses, which results in a substantial drop in capillary conductivity (Ward 1967). Recall that water movement in soils is restricted when a water film around soil particles is absent (Knapp 1978). Thus, under strong conditions of surface evaporation, the capillary conductivity of surface soils can be reduced substantially and the soil may remain moist below these surface layers as further evaporative losses are minimal (Ward 1967). Capillary rise in semi-arid areas (such as the Overberg) can also lead to a strong accumulation of salts near the surface that influence soil and surface water salinity (Bugan, De Clercq & Jovanovic 2010; Ward 1967).

Within unsaturated soils, water moves both vertically and laterally (Weyman 1970), sometimes also referred to as subsurface flow or throughflow. Subsurface flow is a critical component of catchment hydrology. Historically, catchment-level hydrological models associated streamflow peaks with the generation of surface runoff in the drainage basin (Whipkey & Kirkby 1978), but field observations have shown that peaks on a stream's hydrograph can also be attributed to subsurface flow (Rawitz, Engman & Cline 1970; Whipkey & Kirkby 1978). The lateral proportion of soil throughflow can change when vertical permeability is altered within the soil profile. This typically occurs at the soil–bedrock boundary or at another impermeability interface within the soil (Hardie et al. 2012). Subsurface flow can be sufficient to account for flood peaks in some rivers, while can also be a component in other river systems small enough to only help sustain low flows (Whipkey & Kirkby 1978). Regarding the duplex soils present in the Overberg region (Fey 2010), subsurface flow can be greatly influenced by the presence of a discontinuity in the soil profile where a relatively impermeable layer exists lower down (Whipkey & Kirkby 1978). This promotes rapid saturation of the A horizon and a large direct surface runoff flow component, leading to soil erosion from hillslopes.

2.3.1.2 Soil erosion from hillslopes

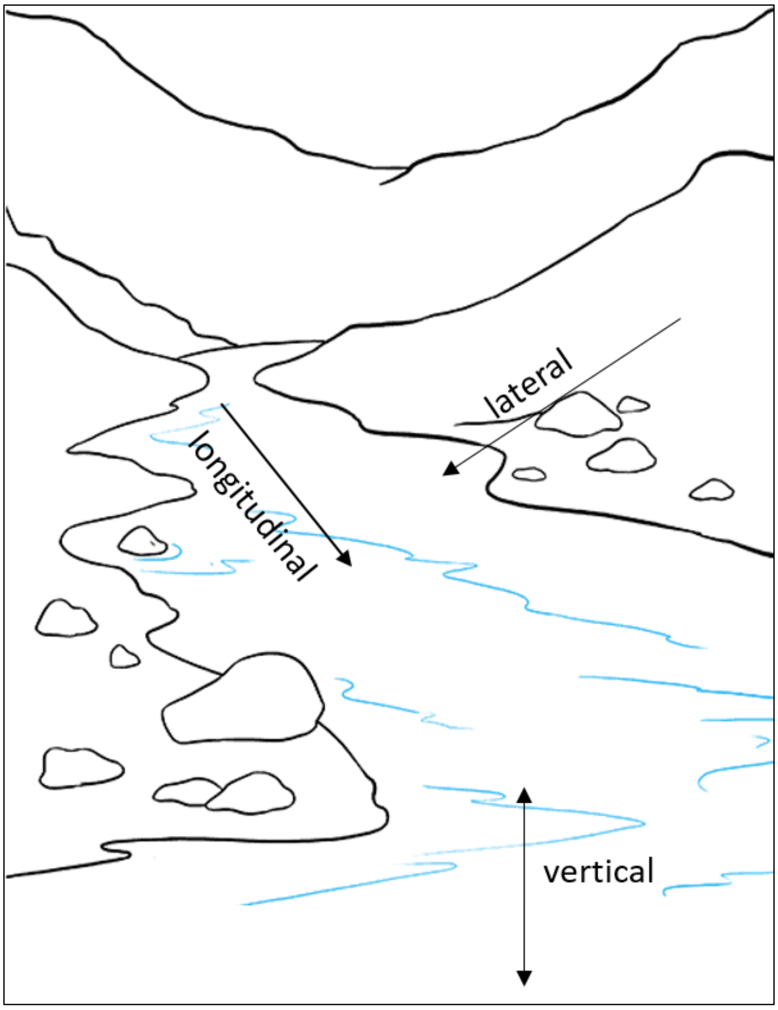
Soil erosion is driven by wind (aeolian) or by water (Issaka & Ashraf 2017) through soil loosening, detachment and transport (Guo et al. 2018). Fluvial erosion is caused by rainsplash or by overland flow (Prats et al. 2019) in the form of either inter-rill erosion or rill erosion (Bryan 2000). The susceptibility of soils to erosion (erodibility) is determined by the soils' characteristics (Bryan 2000). The concept of soil erodibility was first described by Middleton (1930) who outlined that overland flow and particle detachability are the major determinants of erodibility (Bryan 2000). Subsequently, these indices have been reviewed, critiqued and refined (Bryan 1968; Smith & Wischmeier 1962; Song et al. 2005) and several indices have been developed with the intention of creating a universally applicable index of soil erodibility. Various parameters, such as rainfall intensity, landuse, slope and soil properties, are understood to be important determinants of soil erosion (Guo et al. 2018). Consequently, several soil erosion indices have been developed. Noteworthy indices are the universal soil loss equation (USLE) (Wischmeier & Smith 1965), the revised universal soil loss equation (RUSLE) (Renard et al. 1997) the water erosion prediction project (WEPP) (Nearing et al. 1989) and the erosion productivity impact calculator (EPIC) (Williams, Jones & Dyke 1984). The modified universal soil loss equation (MUSLE) (Williams 1975) is used to calculate soil loss due to a rainfall event but deviates slightly from the USLE (Wischmeier & Smith 1965; 1978). Whereas USLE uses rainfall intensity as an estimator of erosive energy (Phuong, Shrestha & Chuong 2017), MUSLE uses the degree of runoff to approximate erosion and sediment yield. MUSLE has been used to estimate sediment erosion in South Africa (Gwapedza et al. 2021) and is a useful tool as it is available in conventional models such as the SWAT (Neitsch et al. 2005), which has been frequently applied in South African studies.

Overland flow and subsurface flow both lead to the delivery of water and hillslope materials such as sediment, salts and nutrients to rivers. The ability of these flow components to do so depends on the hydrological connectivity of various sections (structural units) of the hillslope.

2.3.2 Hydrological connectivity

The concept of connectivity has strong roots in ecological discourse but is being applied increasingly in the hydrological and geomorphological domains (Bracken & Croke 2007; Bracken et al. 2013; Fryirs et al. 2007; Rowntree 2012; Turnbull, Wainwright & Brazier 2008; Ward 1989). Connectivity relates to the integration of functional and structural units in a landscape over a variety of spatial and temporal scales (Harvey 2002; Rowntree 2012). Connectivity thus describes the potential for animals, propagules[8], particulate matter, water and dissolved nutrients to move over a land surface (in hydrological terms typically downslope in a lateral or vertical manner) (Turnbull, Wainwright & Brazier 2008). Figure 2.2 illustrates these movements.

[8] A material (such as a seed or spore) that's acts to propagate a plant, usually by dispersal.



Source: After Rowntree (2012)

Figure 2.2 The concept of hydrological connectivity concept after Rowntree (2012).

Landscape connectivity is defined as being either structural (With, Gardner & Turner 1997) or functional (Baguette & Van Dyck 2007; Tischendorf & Fahrig 2000; With, Gardner & Turner 1997). Structural connectivity refers to a physical connection or link between terrain components such as terrain units on a hillslope or the channel network of a river system as depicted by the longitudinal, lateral and vertical connectivity shown in Figure 2.2 (Rowntree 2012). Functional connectivity is better conceptualised as being associated with processes, that is a measurement of the movement of materials such as water and sediments within a habitat (Pascual-Hortal & Saura 2007; Rowntree 2012).

Hydrological connectivity is directional and is concerned with the connections between hydrological response units (HRUs) down a hillslope catena (Turnbull, Wainwright & Brazier 2008). The concept of connectivity in the field of geomorphology has been often explored in the context of sediment movement within a catchment (Fryirs et al. 2007; Harvey 2002; Rowntree 2012). The connectivity of sediment down channel networks and from hillslopes, shapes and promotes change of a landscape by determining zones of deposition and storage or zones of erosion. Sediment storage zones and sinks are areas of disconnectivity in a catchment.

Fryirs et al. (2007) put forward four factors that lead to disconnectivity in a catchment, namely barriers, blankets, buffers and boosters. Barriers, such as dams and valley constrictions, affect longitudinal connectivity and result in increased disconnectivity along the channel network. Blankets are zones of sediment deposition, where underlying sediments are protected by materials deposited above the sediments. Buffers prevent sediment movement from hillslope to channel and thus reduce lateral connectivity. Boosters (pinch points or valley constrictions) such as gorges are features that may increase disconnectivity. Boosters are widespread phenomena in South African river systems (Rowntree 2012). Boosters are typically straightened river reaches that increase the capacity of a river to move sediment downstream. However, a marked loss of flow energy downstream of these features results in a rapid transition into depositional zones (Fryirs & Brierley 2013).

The connectivity of distinct HRUs on a hillslope affects the delivery of water and sediment downslope (Ocampo, Sivapalan & Oldham 2006). While the connectivity of surface flows is often given greater attention (as they are more visible), the connectivity between hillslopes and riparian zones can be described in terms of both surface (overland) flow and subsurface flow. Subsurface connectivity between upslope areas and downslope areas is determined by the water table lying above a confining layer within the soil (Ocampo, Sivapalan & Oldham 2006; Vidon & Hill 2004). Such subsurface connectivity is treated next.

2.3.3 Subsurface hillslope connectivity

Subsurface flow in hillslopes is responsible for the significant movement of solutes and nutrients and it is the dominant hydrological response in environments such as forested catchments in high-rainfall areas (Blann et al. 2009; Stieglitz et al. 2003). Two preferred hydrologic states predominate temperate landscapes – dry state and wet state (Grayson et al. 1997). A dry state is characterised by vertical fluxes in water, while a wet state is dominated by the lateral movement of water through surface and subsurface flow (Grayson et al. 1997), thus describing the connectivity of the underground system. In areas with strong rainfall seasonality, potential evapotranspiration may far exceed rainfall in dry seasons resulting in the drying of soils and low hydrologic connectivity. Under these conditions, any rain that does occur serves to only wet the topsoil uniformly and is subsequently evapotranspired prior to any substantial lateral redistribution (Grayson et al. 1997). As rainfall increases during the rainy season, wet areas develop where there is substantial convergence caused by catchment terrain. Runoff moves downslope under the influence of gravity, both through overland and subsurface flow, and quickly wets drainage channels.

Lateral connectivity of subsurface flow is dependent on rainfall. In the transitional period between wet and dry states a rise in evapotranspiration and decreased rainfall result in drying soil and diminished lateral flow until it is non-existent (Grayson et al. 1997). Lehmann et al. (2007) have suggested that lateral movement of water only takes place through soil sites with a water table. Thus soil in close proximity to bedrock becomes saturated during a rainfall event and only then does the soil move downslope. Tromp-van Meerveld & McDonnell (2006) found that lateral hillslope flow was greatly limited by whether bedrock depressions had been filled or not. It is suggested that this lateral near-surface flow my take preferential pathways including macropores and bedrock valleys (Lin 2010; Lehmann et al. 2007).

The measurement of the connectivity of HRUs, as well as subsurface flow and surface runoff, in hydrological systems is a challenging task (Beven 2012). There are usually limited numbers of flow gauges and rain gauges in any catchment so that the determining of rainfall–runoff–streamflow relationships requires extrapolation from available spatio-temporal data with the aid of hydrological models.

2.3.4 Brief history of hydrological modelling

Rainfall–runoff modelling dates back to the middle of the 19th century when Thomas Mulvaney first described the rational method (equation) (Beven 2012; Chow, Maidment & Mays 1988; Mulvaney 1851) for determining flood peaks in rivers (Dooge 1986). The rational method predicts the hydrograph peak (Q_p) with the following equation:

$$Qp = C\frac{h_c}{T_c}A \qquad \text{Equation 1}$$

Where Q_p is the hydrograph peak discharge, A is the catchment area, h_c is the critical rainfall, T_c is the concentration time (h) and C is an empirical, scaling (runoff) coefficient that estimates how much rainfall becomes surface runoff (Grimaldi & Petroselli 2014; Piscopia, Petroselli & Grimaldi 2015). The rational method remains the most used equation applied in practical hydrology as it is simple, calculates peak flow and requires relatively little data (Grimaldi & Petroselli 2014). However, Grimaldi & Petroselli (2014) also point out several flaws in the rational method that pertain to the appropriate value for the runoff coefficient. The rational method also has limitations regarding the prediction of extreme events and applications in ungauged catchments (Beven 2012).

Hydrological modelling considers the processes of runoff generation and runoff routing. These processes are not homogenous over an entire catchment, so giving rise to the use of distributed hydrological models (Brirhet & Benaabidate 2016; Tran, De Niel & Willems 2018). While a lumped hydrological model considers sub-basins as single units (often delineated by elevation zones), fully distributed models divide sub-basins into smaller units based on soils, topography and landuse (Brirhet & Benaabidate 2016). Beven (2012) contends that the first attempts to generate a distributed hydrological model were made by Imbeaux (1882) where he worked on flooding in southern France. Imbeaux divided the studied catchment into zones delineated by the travel time (of water) to the river's outlet. Runoff in each zone was calculated for each area and then routed to the catchment outlet to simulate the stream hydrograph. This concept was used by other hydrologists as interest in the field began to grow in the 20th century (Richards 1944; Turner & Bourdoin 1941; Zoch 1934) and the principles are still used in contemporary distributed models (Beven 2012). Subsequently, further methods were developed to model catchment hydrology, such as the unit hydrograph (Edson 1951; Sherman 1932) and the United States Department of Agriculture Soil Conservation Service (SCS) method (McCuen 1982). The unit hydrograph approach does, however, not take into account that some of the streamflow occurs, even when rainfall is absent, due to the baseflow component of the river driven by subsurface flow (Beven 2012). It has therefore become necessary to separate the hydrograph into various flow components, particularly surface flow and baseflow.

The advent of geographical information systems (GIS) databases allows the superimposition of soil, landuse and topographical data to delineate and classify units in a catchment with different functional responses to rainfall inputs (Beven 2012). These HRUs and their topography are used to determine flow direction and distance to the outlet, thus defining the flow routing. Each HRU's runoff is routed to the outlet and is part of the simulated hydrograph (Watson et al. 2019). As a result of these geospatial databases and enhanced computational power, there has been a proliferation of distributed models since the 1990s. Distributed models can separate flow components and facilitate the use of flow pathways to predict sediment and pollution movement, as well as being useful for evaluating the impact of catchment changes (e.g. landuse change) on hydrological and sediment dynamics (Beven 2012; Gwapedza et al. 2021; Pfennig et al. 2009).

Noteworthy distributed hydrological models that have been developed are the Systéme Hydrologique Européen (SHE) model (Abbott et al. 1986; Bathurst 1986), the United Kingdom's Institute of Hydrology distributed model (IHDM) (Calver & Wood 1995), the Australian THALES model (Grayson, Blöschl & Moore 1995) and the integrated hydrologic model (InHM) developed in the United States of America (VanderKwaak & Loague 2001). In South Africa the Jena adaptable modelling system (JAMS) (Krause 2001; Kralisch et al. 2007) has been used to model streamflow components for many catchments (Watson, Kralisch, et al.

2021; Watson et al. 2019; Watson et al. 2018) and specific regional parameter sets have been developed (Watson, Midgley, et al. 2021). This makes the application of the JAMS/J2000 model an appropriate choice for hydrological and sediment modelling in dryland rivers such as the Bot River.

2.3.5 Hydrology of dryland rivers

Dryland environments are characterised by their relatively sparse vegetation cover due to aridity and by a notable seasonal variability in moisture availability (Bull & Kirkby 2002). Certainly, the outstanding dryland environment is the Mediterranean climatic zone (Bull & Kirkby 2002; Suc 1984) which has characteristic summer drought and winter rains. Dryland rivers have been the subject of growing research interest (Nanson, Tooth & Knighton 2002). These river systems have several regional parallels and differences. Regarding climate, many dryland catchments are prone to intense rainfall events, which, coupled with the sparse vegetation, lead to high levels of overland flow and subsequent soil erosion from hillslopes (Bull & Kirkby 2002). Most dryland rivers are located in subtropical regions where they are impacted by the presence of dry, stable air masses that limit the potential for rainfall (Nanson, Tooth & Knighton 2002). Accordingly, high levels of aridity persist in these zones giving rise to the presence of drylands. River hydrology is often impacted by whether the river is exotic (with a major source of water outside the dryland area through which it flows) or endogenic (sourced within the dryland zone itself) (Nanson, Tooth & Knighton 2002). Exotic rivers are often perennial, whereas endogenic are typically ephemeral or intermittent. This is typical of the Bot River catchment where tributaries from the east of the catchment (such as the Swart River – Figure 1.5) are dry in the summer months, while tributaries from the high mountainous area to the west of the catchment contribute flow that makes the Bot largely perennial (with the odd exception of particularly dry years).

Dryland rivers are subject to four types of flood, namely (i) flash flooding, (ii) single-peak floods, (iii) multiple-peak floods and (iv) seasonal flooding (Graf & Lecce 1988). Flash floods and single-peak events are most prevalent in endogenic systems, whereas multiple-peak floods and seasonal floods are more typical of exotic rivers (Nanson, Tooth & Knighton 2002). The hydrograph most commonly associated with dryland rivers (particularly in smaller catchments) is the flash flood which has characteristically steep rising and falling limbs in response to a substantial overland flow component (Nanson, Tooth & Knighton 2002). Dryland rivers can transport substantial amounts of sediment during flood events as both suspended load and bedload (Nanson, Tooth & Knighton 2002).

In many dryland areas streamflows are insufficient to maintain bank and in-channel vegetation growth, hence riparian vegetation in dryland rivers is typically sparse (Nanson, Tooth & Knighton 2002; Tooth & Nanson 2000). Invasion of the riparian zone by alien plants is a common occurrence in dryland rivers (Nanson, Tooth & Knighton 2002). In the Overberg alien invasion is problematic along many of the rivers (Holmes et al. 2020; Mtengwana et al. 2021; Sieben & Reinecke 2008). Invasion by alien plants affects biodiversity, water-resource availability and the provision of ecosystem services across much of South Africa (Holmes et al. 2020; Van Wilgen et al. 2016). In lowland vegetation systems like the Overberg, *Acacia saligna* (Port Jackson) and *Acacia cyclops* (rooikrans) and *Leptospermum laevigatum* (tea tree) are particularly prevalent (Holmes et al. 2020). Acacias are able to create large seedbanks in the soil and often resprout robustly after disturbance by fire or felling (Strydom et al. 2017). Due to the substantial habitat fragmentation in lowland areas and their nearness to cultivated fields, lowland renosterveld areas are more often invaded by alien weed and grass species (Holmes et al. 2020), especially grasses and forbs. Riparian zones tend to be invaded by acacias (*Acacia longifolia* (long-leafed wattle) and *Acacia mearnsii* (black wattle)) which often dominate the river banks and outcompete indigenous species. This results in substantial losses of biodiversity and ecosystem services in the landscape.

2.4 ECOSYSTEM SERVICES

Costanza et al. (2011, p.1) define ecosystem services as the “ecological characteristics, functions or processes that directly or indirectly contribute to human well-being.” As such, ecosystem services are a notion that is largely human-centric in nature (Everard 2017), where humans are the ones who benefit from the services an ecosystem may provide. The use of the concept of ecosystem services in scientific discourse arose in the 1960s, partly to inform the general public on the value and worth of ecosystems and the benefits they provide – thus creating increased awareness of environmental concerns and the growing need for conservation of natural resources and ecosystems (Ehrlich & Ehrlich 1981; Sutherland & Mazeka 2019). The practicality of the concept has since grown to include determining the economic value of the services rendered with many scholars investigating methods for assigning an actual economic value to ecosystem services (Costanza & Daly 1992; Gómez-Baggethun et al. 2010).

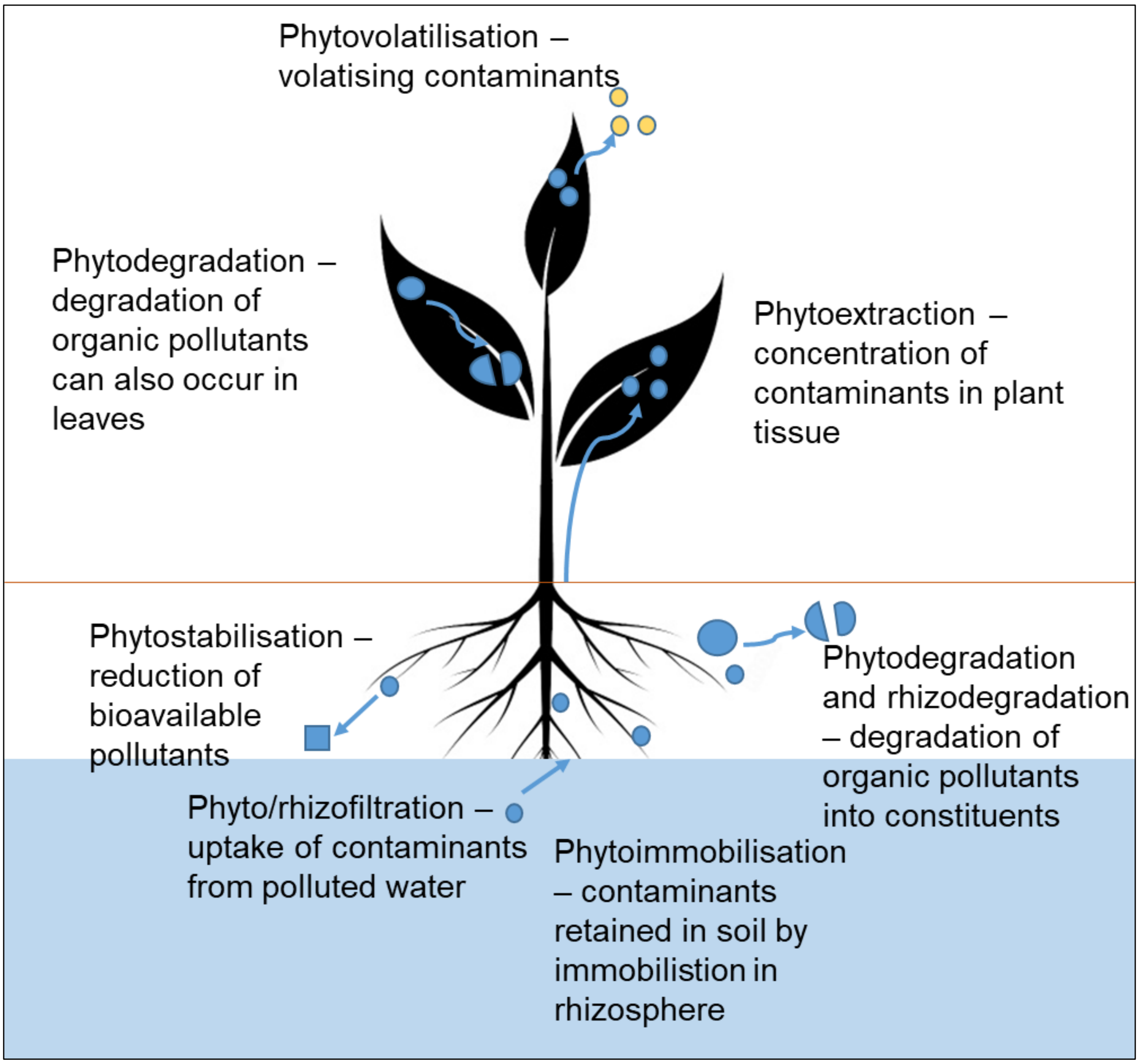
Sutherland & Mazeka (2019) have proposed that the concept should include an appreciation of the available natural resources that aid the functioning of these ecosystems which, in turn, provide services. The concept of ecosystem services is mainly applied in the South African context to promote the conservation of biodiversity for optimal ecosystem functioning (Blignaut et al. 2008; Sutherland & Mazeka 2019; Turpie et al. 2017). South African ecosystems contain high levels of biodiversity, particularly in biodiversity hotspots such as the

CFR (Rebelo et al. 2006). Five critical environmental services have been identified by Egoh et al. (2008), namely surface water supply, flow regulation, soil accumulation, soil retention and carbon storage. However, phytoremediation has also been identified as a valuable ecosystem service (Dickinson et al. 2009). Arthur et al. (2005) have suggested that the use of vegetation may be a low-cost, efficient way to remediate degraded and contaminated systems through the phytoremediation of pollutants historically released into a landscape.

Several key ecosystem services are offered by the natural vegetation in drylands (García-Palacios et al. 2019). The capacity of soils to retain water is a crucial service to dryland agriculture (García-Palacios et al. 2019) and it is often determined by soil structure (Maestre et al. 2016). Soil water retention is typically low where meagre contents of organic matter and coarse soil textures coexist (Lal 2004). Under such conditions nutrient availability in dryland soils is similarly low. The tilling of soils regrettably leads to substantial losses in soil organic content, thus resulting in a loss of this ecosystem service (García-Palacios et al. 2019; Plaza-Bonilla et al. 2015). Two other ecosystem services are carbon accretion and nitrogen retention (García-Palacios et al. 2019). Moreover, indigenous renosterveld fragments increase infiltration of rainfall into soils and act to limit topsoil loss (O'Farrell, Donaldson & Hoffman 2009). Several studies have found that vegetation buffers hold great potential for improving the quality of runoff water from agricultural hillslopes by the process of phytoremediation (Allaire et al. 2015; Lee et al. 2000; Lerch et al. 2017; Krutz et al. 2005).

2.5 PHYTOREMEDIATION

Phytoremediation has been defined as the use of individual or plant assemblages to remediate (remedy) soil or water contaminated with hazardous chemicals (Arthur et al. 2005). Locally and regionally, soils and water may become contaminated by pollutants stemming from natural and/or anthropogenic sources. Phytoremediation processes are best suited for areas which exhibit shallow contamination of <5-m depth and where the contaminants are moderately hydrophobic (tending to repel or fail to mix with water) (Schnoor et al. 1995). These pollutants can be traced to individual points (point sources) or to larger, more diffuse areas (non-point sources) (Arthur et al. 2005). Of particular concern are the contaminants stemming from agricultural activities from where organic and inorganic compounds can enter watercourses. Phytoremediation holds great potential to remedy the contamination of soils, as well as near-surface and surface water, having organic and inorganic compounds in situ (Dosnon-Olette et al. 2011; Schnoor et al. 1995). Phytoremediation comprises multiple processes, namely phytovolatilisation, phytofiltration, rhizofiltration, phytoextraction, phytoimmobilisation, phytostabilisation and phytodegradation and rhizodegradation as shown in Figure 2.3.



Source: After Rigoletto et al. (2020)

Figure 2.3 Conceptual diagram of major phytoremediation processes.

Although physical and chemical processes, such as leaching with surfactants[9] and volatilisation[10] via air vents, may be used to remediate contaminated areas both in situ and ex situ, such methods have been shown to be prohibitively expensive. In agricultural areas nitrates are a common contaminant of waterbodies that lead to excessive nutrient loading and the resultant eutrophication of waterbodies and rivers. Nitrate removal is expensive and methods

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[9] Surfactants are substances that lower the surface tension of the liquid in which they are dissolved.

[10] Volatilisation occurs when a chemical in liquid form is converted to a vapour.

such as reverse osmosis result in highly concentrated wastewater streams (Arthur et al. 2005). In contrast, phytoremediation is presented as a low-cost, efficient mechanism for removing contaminants from degraded waterbodies exhibiting excessive chemical loading.

The rate at which compounds are taken up by plants is determined by the physiochemical properties of each compound. For instance, regarding organic pollutants, moderately hydrophobic chemicals (with log octanol-water coefficients of 0.5 to 3.0) are most likely to be sorbed[11] by root systems and taken up into a plant's tissue (Schnoor et al. 1995). Strongly hydrophobic chemicals bind powerfully to roots and are thus not easily translocated within the plant while more soluble chemicals are typically not well sorbed to roots (Schnoor et al. 1995).

The processes described in Figure 2.3 as well as the efficacy and use of vegetation buffers are described in the following subsections. First phyto or rhizofiltration is considered (Section 2.5.1), followed by the processes of phytoextraction (Section 2.5.2), phytoimmobilisation and stabilisation (Section 2.5.3), phyto and rhizodegradation (Section 2.5.4), phytovolatilisation (Section 2.5.5) and the use of vegetative buffers in phytoremediation (2.5.6).

2.5.1 Phyto- or rhizofiltration

Phytofiltration is a form of phytoremediation that makes use of plants to abstract contaminants from polluted water. In this process compounds are taken up into the biomass of hydroponically cultivated plants via roots in contact with the water (Raskin, Smith & Salt 1997; Gardea-Torresdey et al. 1998). Common applications include the removal of nutrients and metals from industrial waste and agricultural wastewater, while the presence of bacteria in the rhizosphere (the soil surrounding plant roots) is crucial to maximising the rate of contaminant sorption to plants (De Souza et al. 1999). It is the binding mechanisms in the root tissue of plants that enables the effective removal of pollutants from waterbodies. Phytofiltration has been applied in a variety of contexts by using various plant species to bioaccumulate chromium and other heavy metals (Lombi et al. 2001), lead (Sekhar et al. 2004), copper, nickel and zinc (Gardea-Torresdey et al. 1998) and nitrates (Hashemi et al. 2016; Li et al. 2016) among other compounds.

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[11] The process of taking up and holding nutrients and contaminants

2.5.2 Phytoextraction

Phytoextraction is the capability of plants to take up inorganic pollutants (primarily metals) from the soil (Arthur et al. 2005). Some of the inorganic chemicals taken from the soil are used for plant growth, while others have no known function. Ernst (1996) maintains that the degree of phytoremediation achieved by phytoextraction is determined by the level of pollution and the capacity of a plant to absorb and accumulate metals. The mechanism of contaminant uptake is through root systems via an aqueous phase (Arthur et al. 2005). In the process ions in the soil move into the roots simultaneously with the uptake of water by the plant as part of the transpiration process. Additional mechanisms that facilitate chemical uptake are diffusive transport and microbial-assisted transport.

Metal-chelate complexes are important for metal sorption to roots. In this process, metal-deficient plants release chelating[12] agents into the rhizosphere. These agents bind metals, which are subsequently transported through the root cell's plasma membrane via a specific protein for each metal (Arthur et al. 2005). Many plants are metal tolerant, but most do not allow substantial accumulation of metals in the biomass. Those plants that do are termed hyperaccumulators and are able to contain metals at a concentration of > 0.1% (Brooks 1998).

2.5.3 Phytoimmobilisation and phytostabilisation

Phytoimmobilisation is a phytoremediation technology describing a process of nutrient uptake by plants and the later release of these chemicals from decomposing organic material into a geomat or mineral-amended soil (Arthur et al. 2005). Phytostabilisation results when toxic metals are removed from soils through the complexing of metal compounds in plants. Although the chemical itself is not removed from the system, the hazardous nature of the contaminant is reduced (Arthur et al. 2005).

2.5.4 Phyto- and rhizodegradation

Phytodegradation is the process of breaking compounds down into constituents (Arthur et al. 2005). Degradation products are typically less toxic than the original compound (though this is not always the case) and these products are formed either in the rhizosphere or in the actual biomass of the plant. Phytodegradation is thus the transformation of compounds in the plant

[12] Chelation "is a process involving the formation of a heterocyclic ring compound which contains at least one metal cation (or hydrogen ion) in the ring" (Van der Walt & Van Rooyen 1995, p.31).

tissues where enzymes work to transform the contaminant compound into constituents or products that are more readily used or released (Arthur et al. 2005). In contrast, rhizodegradation takes place in the rhizosphere where bacteria and fungi (or even enzymes released by plant roots) are responsible for the transformation of these compounds. Process efficacy is affected greatly by a wide variety of factors, including temperature, pH, soil moisture content and aeration.

Organic pollutants may be remediated through three processes: (i) Direct uptake of contaminants and build-up in plant tissue; (ii) releasing of substances (exudates) and enzymes to enhance microbial activity and subsequent biochemical transformation rates; and (iii) increased mineralisation in the rhizosphere (Schnoor et al. 1995).

2.5.5 Phytovolatilisation

Phytovolatilisation describes the process of separating contaminants into air spaces within plant tissues and the later diffusion of these contaminants into the surrounding air (Limmer & Burken 2016). Various contaminants, ranging from metals and metalloids (De Souza et al. 1999; Guarino et al. 2020) to organic compounds (Baeder-Bederski-Anteda 2003) can be remediated through volatilisation. Plant uptake of pollutants can lead to the volatilisation of these compounds from plant stems and leaves in a process that resembles transpiration (Limmer & Burken 2016). Many phytovolatilised pollutants are somewhat hydrophobic and are therefore able to diffuse through hydrophobic barriers in the plant tissue.

The individual processes described in Section 2.5.1-2.5.5 are all forms of phyoremediation. The research conducted for this study does not consider individual processes of phytoremediation but is concerned with the efficacy of vegetative buffers as a whole to remediate the adverse impacts of pollution. It is likely that only some of these processes are relevant to renosterveld.

2.5.6 Vegetative buffers in phytoremediation

Riparian buffer strips are three-dimensional groupings of vegetation and organisms adjacent to flowing water which denote a distinct zone between the terrestrial and aquatic ecosystems (Dallas & Day 2004). These vegetation zones have demonstrated an ability to limit the transport of nitrogen and phosphorus from agricultural slopes to watercourses by reducing overland flow rates and causing particle retention or by adsorbing nutrients to roots. Buffer strips can include grasses, shrubs and trees or a combination of plants, and they provide an efficient and economical method of removing nutrients, pesticides and suspended solids from overland flow (Borin et al. 2005). Vegetation, especially grasses and shrubs, in buffers filter pollutants from overland flow by increasing surface roughness which, in turn, reduces flow rates of surface runoff and enhances infiltration into the soil substrate (Borin et al. 2005; Liu, Zhang & Zhang 2008). The resulting deposition of sediment and increased infiltration enhances the efficacy of pollutant and nutrient adsorption to soil and plant roots. Nutrients and pollutants are then taken up and phytoremediated by plants in this zone. Vegetation buffers generally show some capacity to reduce sediment and pollutant delivery rates from source to watercourses (Arora, Mickelson & Baker 2003; Arora et al. 1996; Liu, Zhang & Zhang 2008). In the Overberg, many of the alien invasives in the riparian zone are trees (Figure 2.4). This may limit their phytoremediatory potential when compared to indigenous grasses and shrubs found in renosterveld vegetation.

Source: Author

Figure 2.4 Alien trees are common in the Bot River riparian zone.

Various studies have shown the efficacy and use of vegetative buffers as active phytoremediatory zones standing between pollution sources and river systems. Schnoor et al. (1995) demonstrated the potential for vegetative buffers to reduce levels of pollution by planting rows of poplar trees as riparian vegetation – an 8-m-wide buffer strip. The results showed a substantial decrease in nitrate concentrations in surficial (near-surface) groundwater from 50-100 mg/l to <5 mg/l after interception and remediation. Thus the effective management, preservation and restoration of vegetative buffer zones located between agricultural slopes and rivers may assist in the removal of nutrients available from agricultural processes. Phytoremediation of mobilised nutrients by natural vegetation buffers removes dissolved nutrients that would otherwise leach out or make their way into river systems (Lee et al. 2000; Schoumans et al. 2014). Riparian vegetation and meadows are known to function in this manner with many studies having demonstrated the potential of these systems to filter out particulate materials and dissolved nutrients (Stutter, Chardon & Kronvang 2012; Stutter, Langan & Lumsdon 2009; Tanner, Nguyen & Sukias 2005). The presence of riparian vegetation evidently reduces the potential for river or stream contamination from overland flows and subsurface leaching (Lowrance, Leonard & Sheridan 1985).

Phosphorus is not particularly mobile in soils due to low solubility (Shen et al. 2011) and transportation to watercourses primarily occurs due to the erosion of soil from hillslopes where phosphorus is bound strongly to soil particles. Sediment deposition in buffer zones can assist in reducing delivery rates to watercourses and engender a concomitant reduction of contamination from phosphorus (Hoffmann et al. 2009; Schoumans et al. 2014). Riparian buffers have been shown to reduce total phosphorus delivery to watercourses by 41-92% depending on the environmental factors involved (Schoumans et al. 2014). Nitrogen is, however, much more mobile in soils and is easily leached beyond the rhizosphere and root zone. Studies demonstrated the potential of vegetation buffer strips and wetland areas to reduce water pollution from nitrogen (Bouraoui & Grizzetti 2014; Lam, Schmalz & Fohrer 2010). Therefore, by reducing sediment delivery rates to streams and by the phytoremediation of dissolved nutrients, riparian buffer strips can reduce the contamination of surface water from overland and subsurface flows off agricultural hillslopes.

The ability of buffer strips to remove pollutants is determined by the morphology of the buffer itself as well as the specific plant species present in the buffer zone (Liu, Zhang & Zhang 2008). Notably, buffers can act as sediment sinks and limit nutrient movement in the landscape as well as allow plants to take up nutrients from the soil. Seven factors that affect the efficacy of buffer strips are summarised here.

Soil type. Vegetation buffers reduce the flow rates of surface runoff from cultivated fields and promote the infiltration of water into the soil. The reduced flow velocity also aides the deposition of sediments in the buffer. Larger particles settle out quickly – within the first few metres of contact with the buffer – while finer, clay particles remain in suspension and travel further into the buffer strip (Gharabaghi, Rudra & Goel 2006).

Buffer width. If a buffer zone is too narrow it provides limited potential for sediment trapping. Thus wider buffers are essential for the deposition of fine soil particles.

Area ratio. This is the ratio of buffer area to source area (agricultural fields). Areas with higher area ratio buffers (i.e. larger buffers in comparison with total drainage area) exhibit a greater efficacy of sediment deposition and a subsequent reduction in pollutant delivery to surface waters (Arora et al. 1996).

Flow. It is often assumed that surface runoff through a vegetated buffer is laminar[13] throughout. However, it is common for flows to concentrate in certain areas along a hillslope (often resulting in rill and gully erosion). An increase in runoff velocity leads to a substantial decline in buffer efficacy in these areas (Liu, Zhang & Zhang 2008).

Slope. Steeper slopes generate greater rates of surface flow and reduce a buffer's ability to trap sediments due to greater overland flow.

Rainfall intensity. High-intensity rainfalls lead to increased surface runoff and reduced infiltration into soils.

Vegetation. The height and density of vegetation and type of plants in the buffer affect the efficacy of nutrient removal.

Organic material in soils. A further mechanism of nutrient retention occurs through sorption by higher organic matter in soils where densely vegetated buffer strips occur.

Overall, buffer strips have proven to be effective options to mitigate the impacts of agricultural pollution on river systems. Where water quality is impacted by nutrient loading, options such as buffer construction should be explored.

2.6 WATER QUALITY IN SOUTH AFRICAN RIVERS

This section provides an overview of several water-quality related studies that have been conducted (Section 2.6.1), the evaluation of water quality (Section 2.6.2) and the monitoring of water quality (Section 2.6.3) in South Africa.

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[13] Laminar flow is water that flows smoothly over a surface (as opposed to turbulent flow). There are no eddies and cross-currents and the fluid moves roughly parallel to the surface (Van der Walt & Van Rooyen 1995).

2.6.1 Water quality studies in South Africa

Research on water quality in South African freshwaters has focused largely on the salinisation of rivers (Hohls et al. 2002; Van Niekerk, Silberbauer & Hohls 2009); the impact of wastewater treatment works on rivers (Dalu et al. 2019; Mema 2010; Momba, Osode & Sibewu 2006; Olabode, Olorundare & Somerset 2020); eutrophication of reservoirs (Harding 2015; Matthews 2014; Munyati 2015); acid mine drainage (AMD) (Geldenhuis & Bell 1998; McCarthy 2011; Naicker, Cukrowska & McCarthy 2003; Oberholster et al. 2010); and macro-level analysis of nutrient status (De Villiers & Thiart 2007; Griffin 2017; Griffin, Palmer & Scherman 2014). Hohls et al. (2002) have reported that the major water-quality problems in the country relate to pervasive elevated salt levels as well as high fluoride (F) concentrations in many areas. Salinity issues linked to geology are particularly prevalent in the Lower Orange, Fish to Tsitsikamma, Gouritz, Berg and Breede water management areas (WMAs) (De Clercq, Fey & Jovanovic 2009). Elevated levels of nutrients in rivers are perhaps the major water quality problem in South Africa, as discussed later in this section.

Surface water chemistry in South Africa is greatly influenced by chemical weathering processes, chloride salinisation and pollution from sulphates (Huizenga 2011). Waters in the Western Cape province are typically characterised by a low pH, especially in mountainous regions, due to lack of underlying carbonate rocks and the presence of acidic soils associated with sandstone formations of the Table Mountain Group (Rebelo et al. 2006). Concomitant with this is the release of humic acids from fynbos vegetation which further lowers pH. Higher chloride levels in lowland Western Cape rivers is attributable to the presence of connate salts in soils (De Clercq, Fey & Jovanovic 2009; Du Plessis & Van Veelen 1991) and groundwater, while elevated sulphate levels are associated with the use of gypsum-containing fertilisers (Huizenga 2011). Concerning eutrophication and elevated sulphate levels in rivers, there is a link between landuse and water quality as evidenced by the prevalence and influence of fertilisers in South Africa as a whole (De Villiers & Thiart 2007; Griffin 2017).

Freshwater pollution from agricultural sources and the resulting eutrophication of river systems are serious concerns in South Africa (De Villiers & Thiart 2007; Harding 2015; Van Ginkel 2011). Long-term data on water quality in the country reveals that the nutrient levels (as a result of high nutrient inputs) in 95% of the largest river catchments in South Africa exceed the recommended water-quality guidelines for aquatic plant life (De Villiers & Thiart 2007). Nitrogen and phosphorus inputs are largely sourced by surface runoff and the leaching of nitrogen and phosphorus from fertilised soils. Inorganic nitrogen levels in South African rivers have varied by a factor of ten between 1985 and 2017 (Griffin 2017). Inorganic nitrogen concentrations in South African rivers had a median value of 0.20 mg N/l during the 1980s and these had dropped to 0.02 mg N/l in 2010 (Griffin 2017).

Dissolved phosphate levels in South Africa's rivers increased from 0.016 mg P/l in 1985 to a maximum of 0.036 mg P/l in 2007 but have shown a decline since 2008 (Griffin 2017). This decrease has led to contemporary phosphorus levels comparable to those of the 1980s. The trend has been attributed to a slight reduction in fertiliser application in 2008, improvements to wastewater treatment facilities across the country and the fact that a major manufacturer of washing detergent in South Africa removed builder phosphorus from its products in 2010 (Griffin 2017). Despite the overall decline in nitrogen and phosphorus levels in South African rivers, substantial seasonal differences in rivers still occur. Seasonal fluxes of nitrogen and phosphorus levels in rivers are controlled by rainfall and by the degree of agricultural transformation in a catchment. Typically, nitrogen levels in rivers are higher after fertilisers have been applied to agricultural fields, which is done during the growth cycle of individual crops. Nitrogen is thus readily available for leaching and movement during rainfall events. In seasons where soils are fallow and fertilisers are not applied, nitrogen transport to rivers is lower. In cases where more than 20% of a catchment's area has been transformed to agricultural landuse, high nitrogen and phosphorus flux values exist (De Villiers & Thiart 2007), although these values are subject to inter-annual and long-term variability. Overall, however, limited information exists for smaller rivers and there is a great need to monitor these systems and evaluate their water quality.

2.6.2 Evaluating water quality and key chemical constituents

The various methods for evaluating the quality of water can be assigned to two major categories, namely bioassessment (including macro and microbiological assessment) and physico-chemical evaluation. Bioassessment is founded on the premise that the responses and health of biota are affected by water quality, thus changes in water quality affect biotic communities as they reflect cumulative impacts over time. The use of biotic communities in assessments of ecosystem health is well recognised (Dallas 1997; Ollis et al. 2006). A multitude of bioassessment methods and indices exist, as described by Ollis et al. (2006), but the principal method used in South Africa is the South African scoring system (SASS) (Dickens & Graham 2002) for macroinvertebrates. On the other hand physico-chemical assessment of water involves measuring the levels of a range of physical and chemical parameters and evaluating them against a set of acceptable standards (Fernández, Ramírez & Solano 2012). There are many water quality indices used globally and different standards are deemed appropriate for various water uses. In South Africa the main water-quality standards used are the South African Bureau of Standards (2015) South African National Standard (SANS) 241 guidelines for safe drinking water and the South African water-quality guidelines for domestic, recreational, industrial, irrigation, livestock aquaculture and aquatic ecosystems (e.g. Department of Water Affairs and Forestry 1996a; 1996b).

This study is concerned with the impact of agriculture on water quality in relation to salinisation and nutrient loading of rivers from agriculture. It has been established that in South Africa agriculture affects the salinisation of soils and rivers significantly (De Clercq, Fey & Jovanovic 2009). The primary focus in the physico-chemical state of the Bot River gives specific attention to salts such as chloride and nutrients nitrogen and phosphorus. Chloride is a highly soluble ion commonly found in freshwater systems under natural conditions (Dugan et al. 2017; Verma & Ratan 2020). Chloride levels in freshwaters typically vary seasonally and inter-annually as climatic conditions change, but they usually remain relatively low and are thus non-threatening. Disproportionate exposure to chlorides may damage or poison aquatic organisms (Verma & Ratan 2020), with the threshold for aquatic life to chronic chloride exposure defined as 230 mg/l (Dugan et al. 2017). Other sources define 120 mg/l for long-term exposure and 640 mg/l for acute, short-term exposure as the thresholds for aquatic ecosystems (Canadian Council of Ministers of the Environment 2011).

Inorganic nitrogen (the major nitrogen components being ammonium, NH_4^+, nitrite, NO_2^-, and nitrate, NO_3^-) is typically present in waterbodies. Nitrogen can be available either in dissolved form or when adsorbed[14] onto suspended particulate matter for uptake by algae and river vegetation (Department of Water Affairs and Forestry 1996b). NH_4^+ is a reduced form of inorganic nitrogen and its abundance is largely controlled by temperature and pH. NO_2^- is the inorganic intermediate form of nitrogen while NO_3^- is the end oxidation product of NH_4^+ (Department of Water Affairs and Forestry 1996b). Owing to the stability of nitrate it is often more abundant than nitrite in aquatic systems although, due to their interconversion, both nitrite and nitrate are frequently considered together. In South African conditions inorganic nitrogen levels in natural, aerobic surface waters are typically <0.5 mg N/l, increasing to >10 mg N/l in highly enriched waters (Department of Water Affairs and Forestry 1996b). In terms of a target water quality range (TWQR) for inorganic nitrogen, the Department of Water Affairs and

[14] "The surface retention of solid, liquid or gas molecules or of ions by a liquid, as opposed to absorption, the penetration of substances into the bulk of the solid or liquid" (Van der Walt & Van Rooyen 1995, p.4).

Forestry (1996b) suggests that (a) concentrations ought not to change by >15% from those of local unimpacted conditions; (b) the state of the waterbody should not increase beyond its present trophic level; and (c) the degree and frequency of natural fluctuations in inorganic nitrogen concentrations should not be altered. Trophic conditions for rivers are defined as oligotrophic (<0.5 mg N/l), mesotrophic (0.5-2.5 mg N/l), eutrophic (2.5-10 mg N/) and hypertrophic (>10 mg N/l) under summer conditions (Department of Water Affairs and Forestry 1996b). For the conservation of aquatic animals TWQ should not exceed 80-350 µg NO_2-N/l and 2000-3600 µg NO_3-N/l (Camargo, Alonso & Salamanca 2005; De Villiers & Thiart 2007), while the suggested concentrations of dissolved inorganic nitrogen for eutrophication prevention are lower than these (De Villiers & Thiart 2007). The Canadian water quality guidelines for nitrate (Canadian Council of Ministers of the Environment 2012) prescribe that freshwater concentrations should not exceed 3 mg NO_3-N/l for long-term exposure or 124 mg NO_3-N/l for short-term exposure. De Villiers & Thiart (2007) define elevated levels of inorganic nitrogen in South African rivers to be $NO_{[x]}$ >400 µg N/l and that seasonal nutrient profiles indicate these conditions for at least five months a year in many rivers in the country.

Phosphorus is an essential macronutrient for a wide variety of living organisms and is understood to be the primary nutrient controlling eutrophication in aquatic ecosystems (Department of Water Affairs and Forestry 1996b). In South Africa phosphorus is rarely found in high concentrations where surface waters remain unimpacted. Phosphorus concentrations from 10-50 µg/L are common in South African rivers, although concentrations in unimpacted waters can be as low as 1 µg/L of soluble inorganic phosphorus (Department of Water Affairs and Forestry 1996b). De Villiers & Thiart (2007) suggest that the maximum PO_4^{3-} level for aquatic animal life be set at 100 µg P/L and that concentrations of >30 µg total P/l are considered conducive to eutrophication.

2.6.3 Monitoring water quality in South Africa

In order to evaluate how water chemistry and river health change over time, the long-term monitoring of sites needs to be conducted. Four of South Africa’s many monitoring programmes are briefly described below.

2.6.3.1 National chemical monitoring programme

The national chemical monitoring programme (NCMP) comprises a number of past national and local monitoring programmes, many of which are now defunct (Van Niekerk 2004). The monitoring network consists of locations at existing Department of Water and Sanitation (DWS) gauging stations where data on basic salts are collected. In 1999 water quality data were available at 2068 sample monitoring stations in the country (Huizenga 2011). However, by 2009 monitoring by the NCMP was reduced to 40 primary sites and 660 secondary sites, about half of which were sampled at approximately two-week intervals (Griffin, Palmer & Scherman 2014). Data from the programme are therefore limited in terms of the frequency of data collection and the number of sample sites for each river, with multiple catchments being unmonitored. In the Bot River there is one DWS water quality station located at the Roode Heuwel weir (G4H014-A01) near the town of Bot River.

2.6.3.2 National eutrophication monitoring programme

Prior to 2002 eutrophication monitoring in South Africa was mainly conducted in an impromptu fashion with broad research projects resulting in long-term data sets such as those of Toerien, Hyman & Bruwer (1975) and Van Ginkel et al. (2000). The national eutrophication monitoring programme (NEMP) (Department of Water Affairs and Forestry 2002) was the result of the formalisation of these monitoring programmes with a focus on reservoirs, with more than 80 reservoirs being monitored countrywide. This programme is not applicable to this study as this dissertation is not concerned with impoundments, but rather with the nutrient levels of flowing river systems in the Overberg.

2.6.3.3 The South African river health programme

The South African river health programme (RHP) was effected in 1994 with the purpose of monitoring the ecological state of rivers in the country (Ollis et al. 2006; Roux 1997). The RHP uses the biomonitoring of fish, aquatic macroinvertebrates and riparian vegetation, coupled with abiotic factors such as habitat integrity and geomorphology to evaluate a river’s ecological health (Dallas 2005). The RHP was replaced in 2016 by the river ecostatus monitoring programme (REMP) (Department of Water and Sanitation 2019). This study does not consider bioassessment and river geomorphology and so the RHP/REMP is not useful for comparison.

2.6.3.4 The national microbial monitoring programme

The national microbial monitoring programme (NMMP) was designed in 1994 and pilot-scale monitoring was implemented from 1997 (National Microbial Monitoring Programme 1999) to monitor areas at high risk of potential faecal contamination that could pose major health risks to water users (Van Niekerk 2004; Venter, Kühn & Harris 1998). The programme aims to monitor *Escherichia coli* and other faecal coliforms at multiple locations across the country (Department of Water and Sanitation 2022). This study does not concern *E.Coli* contamination of the Bot River and so the NMMP is not used for comparative analysis.

Overall, rivers in South Africa have been demonstrably affected by landuse change. Salinisation and nutrient loading in rivers are two major concerns, resulting in the need for robust monitoring networks across the country. Unfortunately, monetary constraints limit the scope of valuable monitoring programmes such as the NCMP so necessitating the study of under and unmonitored catchments.

2.7 CONCLUSION

There is a large body of literature detailing dryland environments, their hydrological functioning and processes, and the value of these areas. A common and unambiguous message that this literature highlights is the need for further research and that drylands are often overlooked as settings and foci for empirical study (Darkoh 2018; Schimel 2010). Hydrological research and the development of perceptual and conceptual hydrological models date back to the 1800s (Mulvaney 1851) with the seminal contribution being Robert Horton's concept of infiltration rates (Horton 1933) that affect the partitioning of rainfall into the key components of surface runoff and subsurface flow that were crucial to the later development of hydrological models. Several noteworthy textbooks, for example Ward (1967), Kirkby (1978) and Beven (2012) detail the influence of parameters such as rainfall, landuse, soil type and topography on streamflow and soil erodibility. A search of the literature revealed that not much is understood of the hydrology and sediment dynamics of Overberg rivers (Le Maitre et al. 2007). The discovery and review of several hydrological models such as SHE (Abbott et al. 1986) and IHDM (Calver & Wood 1995) informed the choice of JAMS/J2000 (Krause 2001) as the modelling system appropriate to this intended study by virtue of its effective application in other local catchments (Watson et al. 2019) and the usefulness of regional parameter sets that have been generated (Watson, Midgley, et al. 2021). The several erosion models reviewed, such as the USLE (Wischmeier & Smith 1965), led to the MUSLE (Williams 1975) being chosen for evaluating soil erosion in the Overberg region as per Objective 2.

Furthermore, the review made it clear that the ecosystem services that dryland areas provide have been gaining increasing attention, particularly in the renosterveld (Bugan, De Clercq & Jovanovic 2010; O'Farrell, Donaldson & Hoffman 2009; Reyers et al. 2009). The review helped to describe the renosterveld landscape within this dryland context and highlighted the biodiversity and ecosystem services this dryland ecosystem hosts (Cowling 1983; Maestre, Salguero-Gómez & Quero 2012; Rebelo et al. 2006). It became evident that renosterveld undoubtedly offers potential value in its ability to mitigate salinisation of local soils (De Clercq, Fey & Jovanovic 2009), sequester carbon[15] at a substantially higher level than active agricultural fields (Mills et al. 2013) and increase infiltration of rainfall into soils, thus reducing erosion and topsoil loss (O'Farrell, Donaldson & Hoffman 2009). The reviewing of the literature uncovered an ecosystem service in the renosterveld that has not received much scholarly attention, namely the potential phytoremediation of nutrients from agricultural hillslopes by vegetation fragments and remnants downslope. The phytoremediatory potential of individual plant species and assemblages has been researched in many other environments globally but the potential for renosterveld to remediate polluted waters has not yet been confirmed, although evidence from laboratory experiments has suggested that renosterveld patches may be able to do so (Jacklin, Brink & De Waal 2019; 2020). However, those experiments focused on riparian species and not on renosterveld vegetation that occurs outside of the riparian zone. Moreover, the results presented by Jacklin, Brink & De Waal (2020) have not been validated in the field, so justifying the necessity of exploring their potential as per Objective 4.

The impacts of agriculture on the renosterveld landscape are clear and ubiquitous, but they are poorly defined regarding the hydrological impacts and nutrient dynamics in these systems. Thus the impacts of agricultural expansion on the hydrology, sediment dynamics and nutrient levels of river catchments in the Overberg demand quantification to be better understood. The overview of water quality evaluation and monitoring in South Africa (Section 2.6) illuminated the selection of evaluation criteria for water health in the Bot River (Objective 3). This includes the criteria described by the Department of Water Affairs and Forestry (1996b) and De Villiers & Thiart (2007) metrics that have been designed for and tested in South African river systems. Key contributions on South African freshwater quality at macro scale are the journal articles of De Villiers & Thiart (2007), Griffin (2017) and Griffin, Palmer & Scherman (2014), but the absence of data at finer spatio-temporal scales was noted as a rationale for this study. Attention now turns to the filling of these research gaps in the following four chapters.

[15] To capture and store atmospheric carbon in terrestrial sinks (Lal 2008).

CHAPTER 3: LANDSCAPE LEVEL CHANGES DRIVING HYDROLOGICAL RESPONSES IN THE OVERBERG

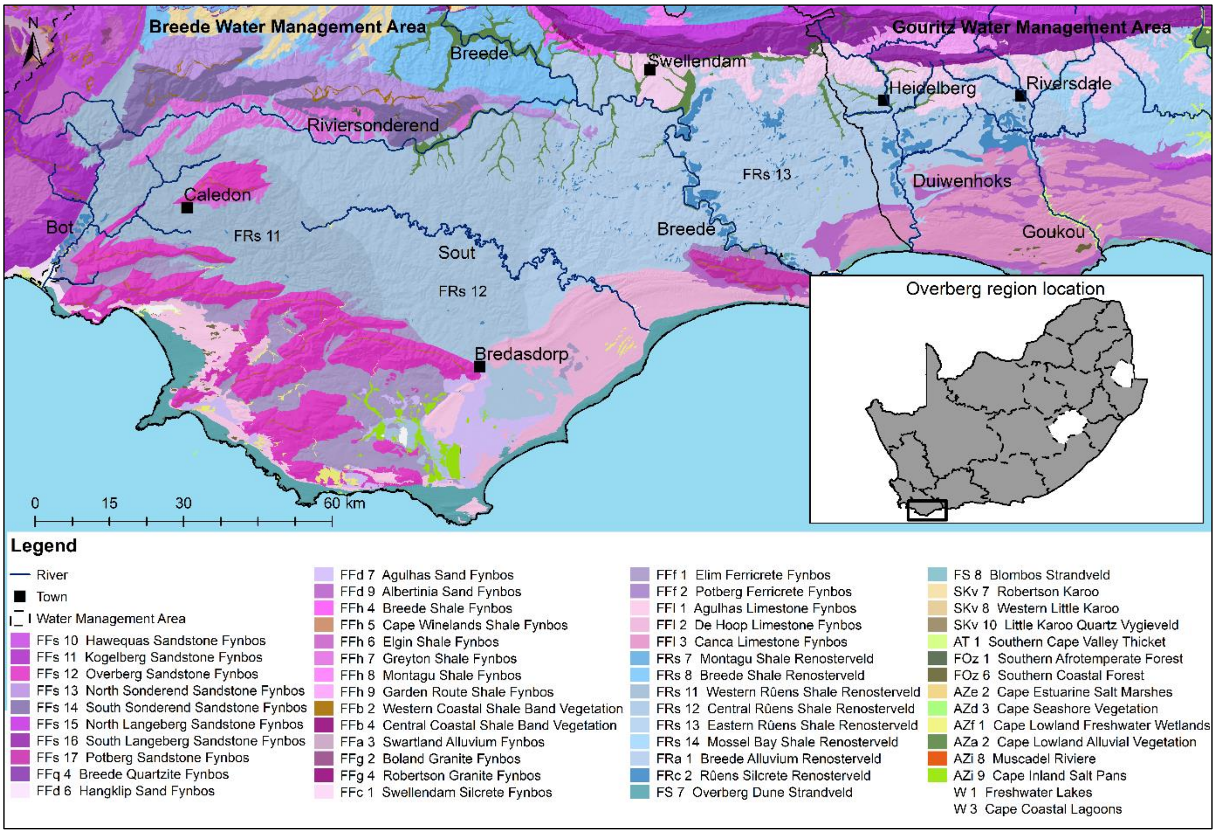
3.1 INTRODUCTION

In this chapter the impact of landuse change, from renosterveld vegetation to cultivated fields, has had on landscape level hydrology in the Overberg region is explored. Drainage basins (or river catchments) are the functional units employed by geographers for examining the interactions between physical, ecological and human processes (Aspinall & Pearson 2000) and they are often used as a reference for hydrological analyses. A drainage basin is a fluvial system that is connected internally by a network of channels and it is delineated by a watershed boundary which separates surface flow between one hydrologic system and another (Fryirs & Brierley 2013). Catchments are characterised by several factors, namely (1) morphological factors relating to the topography of the catchment and the form and extent of the channel network; (2) soil properties that affect infiltration and the moisture relations in catchments; (3) structural factors (geology) that dictate groundwater conditions and the hydrology of the catchment, (4) land cover such as vegetation; and (5) climatic and hydrological factors such as humidity, rainfall and evaporation (Horton 1932). These properties all impact the conversion of precipitation inputs into streamflow. Therefore, catchments that exhibit similar catchment characteristics (physiographic similarity) should have comparable hydrological responses to meteorological inputs. Where the morphological, soil, geologic, landuse and climatic factors change over time it is to be expected that hydrological responses will change too (Aboelnour, Gitau & Engel 2019; Dwarakish & Ganasri 2015; Tomer & Schilling 2009).

The determination of physiographic similarity for the hydrological application of catchment regionalisation has long been used to extrapolate data from areas where a data record exists to those where there is none (Nathan & McMahon 1990). Various techniques have been applied to determine which catchments behave in a similar hydrological manner. Typically, hydrological similarity is determined by likeness between two or more catchments' physical characteristics (Ilorme & Griffis 2013). These characteristics can be used to delineate regions of similarity so that hydrological information can be transferred from a known gauged catchment to others where no gauged data exist. Among these physical characteristics, terrain and topography are crucial elements in the generation of overland flow (Mashimbye, De Clercq & Van Niekerk 2014). Traditional investigations into slope components have relied on the study of topographic maps and aerial photography as well as the making of field observations (Drăguţ & Blaschke 2006), but these methods are now neither time efficient nor cost effective. Digital elevation models (DEMs) and geographic information system (GIS) software are presently used to delineate slope morphological units, hydrological connectivity and landuse homogeneity (Mashimbye, De Clercq & Van Niekerk 2014). DEMs are useful for delineating river catchments and watersheds, while slope elements such as elevation, profile, curvature and gradient can be used to determine relative homogeneity of topography and slope segments over large areas (Drăguţ & Blaschke 2006). Where sufficient data are available this method allows for a rapid and cost effective identification of physiographically similar areas (catchments) within a broader landscape. The interpretation and understanding of these catchment areas and their place within the overall landscape context can be greatly enhanced through the use of GIS software to integrate and analyse multiple data sets.

The use of a river catchment as a geographical unit has a wide range of applications (Maherry et al. 2013). Concerning landscape analysis, environmental and ecological assessment, catchments have long been used as optimal units for analysis and management (Aspinall & Pearson 2000; Haslam 1995), much of the research being based on catchment areas as study sites. A catchment is simply defined as an area that drains water to a central point along a channel network based on topography (Wagener et al. 2007). Catchment units are useful for analyses as they integrate physical, environmental and human processes (which produce stream flow) within defined areas (Wagener, Wheater & Gupta 2004). Catchments can be delineated at various scales (typically from primary through to quaternary level). Large data sets are often associated with these nested hierarchical river catchments (Maherry et al. 2013). The use of quaternary catchments in environmental and hydrological assessments has limits and this has emphasised the need for subquaternary-scale information. Quaternary catchments represent large geographical areas where substantial variations in topography and the physical environment exist. Thus, the delineation of smaller catchments can assist in promoting a more nuanced understanding of landscapes and their hydrological dynamics. Furthermore, the determination of physiographic similarity between these smaller catchments can have applications in landscape analysis and interpretation as well as potential approaches to catchment regionalisation, particularly because hierarchical systems are recognised as being functional units within a larger whole (O’Neill, Johnson & King 1989).

The purpose of this chapter is to demonstrate the effect of landscape transformation on hydrology, by using catchment physiographic similarity in the Overberg, Western Cape (Figure 3.1) as a proxy for hydrological change under natural to transformed landuse.



Source: Mucina, Rutherford & Powrie (2005)

Figure 3.1 Vegetation types in the Overberg region.

This purpose is achieved by two analyses, each one considering different catchment conditions. The first analysis does not take contemporary landuse into account, rather it gives an approximation of similarity under 'natural', undisturbed vegetative conditions that assumes relatively homogenous vegetation across the landscape for each vegetation type. Nine landuse categories are added as variables in the second analysis to depict modern catchment similarity. A comparison of catchment similarity under different landuses can be used as an indicator of landscape level changes in hydrology due to landuse change.

3.2 METHODS AND DATA

The approach for determining catchment physiographic similarity involved delineating smaller (quinary) catchments in the Overberg renosterveld (Section 3.2.1) and using a cluster analysis (Ward 1963) based on several physiographic features for each catchment (Section 3.2.2).

3.2.1 Catchment delineation

Historically, the delineation of river catchments has been achieved manually. These methods were often subjective in nature as well as time consuming and prone to errors (Berelson, Caffrey & Hamerlinck 2004; Maherry et al. 2013). In the South African context, the delineation of secondary and tertiary catchments was initially undertaken in the 1960s by the then Department of Water Affairs and Forestry. These exercises made use of contour lines and spot heights from hard copies of topographic maps (Midgley, Pitman & Middleton 1994). GIS software now makes the delineation quicker and removes the potential for human error. Quinary catchment delineation has been undertaken for the whole of South Africa (Maherry et al. 2013) using a 50-m-resolution DEM. For the purposes of this project, smaller, subquaternary catchments were created in the Rûens Shale Renosterveld vegetation zones using the much higher resolution (5 m) Stellenbosch University digital elevation model (SUDEM)[16] (GeoSmart Space 2019) and the ArcHydro extension in ArcMap 10.3.1. The delineation performed by Maherry et al. (2013) used rules including that every quinary should contain a 1:500 000 river segment (the stretch of river from the river's source to the next tributary, or from one tributary to another). The analysis using the 5 m SUDEM allowed for the delineation of smaller catchments based on a higher accuracy and resolution DEM. When overlaying the two quinary products, there were many differences between the delineated catchments. Furthermore, the quinary catchment product by Maherry et al. (2013) had several catchments with straight catchment boundaries, which are inaccuracies in their product. Thus a new delineation was required for the present study.

A 20-km buffer was generated on the spatial extent of the Rûens Shale Renosterveld (see Figure 3.1) so as to ensure that no delineated subcatchments in the renosterveld would have artificial watersheds. Sinks in the 5-m SUDEM were filled using the ArcHydro extension. Flow direction (D8 method for flow direction determination) and flow accumulation models were then calculated and a stream network was defined using the 'Stream Definition' tool in the same software. This step required that a threshold be set to initiate where a stream would be defined, that is which cells in the raster data represent a stream and which do not. This threshold value relates to the previously calculated flow accumulation, where flow accumulation was defined as the number of cells upslope (in the raster data) that flow into an individual cell. Thus, the number of cells entered in this step affected the density of the delineated stream network. A default value was calculated by ArcHydro to represent 1% of the maximum flow accumulation for the area, although other values can be selected. For this study values ranging from 10 000

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[16] The 5 m SUDEM was interpolated from contours and spot heights data extracted from 1:10 000 and 1:50 000 topographical maps. The 30 m Shuttle Radar Topographic Mission (SRTM) DEM was fused with the interpolated SUDEM in such a manner that accuracy was maximised by preferentially weighting the SRTM dataset in areas of moderate and low slopes where contour density is low. The mean vertical accuracy of the SUDEM is 1.77 m (Van Niekerk 2015).

to 200 000 (in ascending 10 000-cell intervals) were entered in the stream definition phase. A stream grid was subsequently generated where all cells that exceeded the threshold value have a value of "1", while all other cells contain no data. From this layer small catchments were delineated for the region. On visual inspection of the delineated catchments it was decided that a stream definition threshold of 100 000 created hydrological units across the entire study area at an appropriate scale for this study. This is a broad-brush approach to catchment delineation with one setting applied across the entire region. It must be noted that the processing of such a high-resolution DEM over a large area demanded a substantial amount of time and computing power. Basic calculations in ArcMap were conducted to determine the areas and perimeters of the delineated catchment polygons.

3.2.2 Determination of similarity of delineated catchments

Regionalisation of river catchments is common practice in the field of hydrology. The division of a study area into homogenous subcatchments enables data records to be extrapolated more accurately and the confidence of streamflow predictions to be greater (Nathan & McMahon 1990). Homogenous subcatchments have long been defined by many researchers through the use of residuals resulting from a regression equation (Nathan & McMahon 1990; Tasker 1982). The delineation of these subregion boundaries was largely subjective as adjoining regions were created by manual generalisation on a map (Laaha & Blöschl 2006). Subsequently, multivariate techniques such as cluster analysis have been used as more objective measures of catchment similarity (Acreman & Sinclair 1986). This means that subcatchments do not have to be geographically contiguous to form part of the same group (Nathan & McMahon 1990). Common methods for grouping subcatchments include the aforementioned residual pattern approach (Laaha & Blöschl 2006), weighted cluster analysis (Nathan & McMahon 1990), regression trees (Breiman et al. 1984) and the seasonality of low flows technique (Laaha & Blöschl 2006).

Parajka, Merz & Blöschl (2005) suggest that Kriging and similarity approaches to regionalisation perform best. The similarity approach transposes a set of model parameters from one catchment to another that is closest in terms of its physiographic characteristics. With widespread spatial data now available, it is becoming easier to perform such analyses using GIS software and tools. This study adopted a similarity approach and performed a cluster analysis to group physiographically similar subcatchments (Nathan & McMahon 1990).

Despite its usefulness, there are some problems associated with cluster analyses. The selection of variables to determine catchment homogeneity will affect the catchment groupings, while any variable can generate clusters. The selection of variables and any potential weighting thereof are therefore of critical importance. Furthermore, the similarity measure in cluster analysis is principally affected by the scales and units of measurement of the variables, where larger numbers can contribute more to the catchment similarity than smaller numbers. In this analysis many catchments were similar with respect to certain variables but differed substantially in others. Indeed, in some variables the range of values was extremely broad so that, based on an inspection of the data, log transformations were performed on some of the variables. The chosen variables derived from Nathan & McMahon (1990) are listed in Table 3.1.

Table 3.1 Catchment variables used for cluster analysis of catchments in the Overberg region.

Variable	**Log transformed**	**Units**	**Source**
Catchment area	Yes	km^2	SUDEM
Catchment perimeter	Yes	km	SUDEM
Catchment shape parameter (area/perimeter^2)	No		SUDEM
Mean slope	Yes	Degrees	SUDEM
Slope curvature	No		SUDEM
Elevation range	Yes	m	SUDEM
Mean soil depth	Yes	mm	Van Niekerk (2007)
Mean clay content	No	%	Van Niekerk (2007)
Mean annual precipitation	Yes	mm/a	Van Niekerk & Joubert (2011)
Maximum flow accumulation	Yes	No. of pixels (m^2)	SUDEM
Stream length	Yes	No. of pixels (m)	SUDEM
Landuse "Cultivated"	No	%	GeoTerraImage (2015)
Landuse "Water"	Yes	%	GeoTerraImage (2015)
Landuse "Wetland"	Yes	%	GeoTerraImage (2015)
Landuse "Bare ground"	Yes	%	GeoTerraImage (2015)
Landuse "Forest"	Yes	%	GeoTerraImage (2015)
Landuse "Shrub & grasses"	Yes	%	GeoTerraImage (2015)
Landuse "Thicket & woodland"	Yes	%	GeoTerraImage (2015)
Landuse "Urban area"	Yes	%	GeoTerraImage (2015)
Landuse "Other"	Yes	%	GeoTerraImage (2015)
Land type count	No		Van Niekerk (2007)

Source: Adapted from Nathan & McMahon (1990)

Zonal statistics were calculated for each polygon based on several data sets. Detailed *soil data* is lacking (and not widely available) in South Africa where the only soil survey conducted for the entire study area is a land type survey, completed in 2002 and published as a series of memoirs at a scale of 1:250 000 (Agricultural Research Council 2016; Patterson 2005). Although not ideal for spatial analysis, this represents the highest quality soil data available at present. Soil characteristics (soil depth and mean clay content) were obtained from a digitised version of these land types where effective soil depths and clay content were calculated using a soil property extraction process for each land type (Van Niekerk 2008).

Climate data (particularly mean annual precipitation) were obtained from an interpolated, high-resolution climate surface for the Western Cape (Van Niekerk & Joubert 2011), while *landuse* raster data sets from the South African national land cover data were used (GeoTerraImage 2015). The national land cover data contains 72 classes based on 30x30-m raster cells. These classes were merged according to the response each class would have on runoff, to form nine broad classes (for example, shrubland, low fynbos, natural or semi-natural grassland and planted grassland were merged into one class namely shrub and grasses that quite likely exhibits a similar impact on surface runoff based on the vegetation structure). The use of GIS software enabled the overlaying and synthesis of these diverse data sources as well as the calculation of catchment properties.

Little is known about the differences in vegetation structure between Western, Central and Eastern Rûens Shale Renosterveld (Rebelo et al. 2006). Our understanding of the effects of vegetation structure on rainfall–runoff is therefore limited. Delineated catchments and their properties (Table 3.1) were grouped by vegetation region to negate the potential effects of structure differences on the determination of hydrological similarity of subcatchments (see Figure 3.2). A hierarchical, agglomerative tree-clustering method was performed on the three data sets using Ward's method (Ward 1963) in Statistica 13.5. Under Ward's method, each data point is considered its own cluster that is then combined with the nearest cluster in feature space to form a new cluster. The key operation is thus the continued combination of the two nearest clusters until the resulting dendrogram leaves one whole cluster encompassing the entire data set. On inspection of the dendrograms, cluster numbers were set for each vegetation type. Cluster membership for each catchment was determined and pinned to each polygon ID. Cluster membership was then used to spatially represent clusters on a map of the area to display regions of catchment similarity that predict similar runoff responses in these smaller catchments (Parajka, Merz & Blöschl 2005).

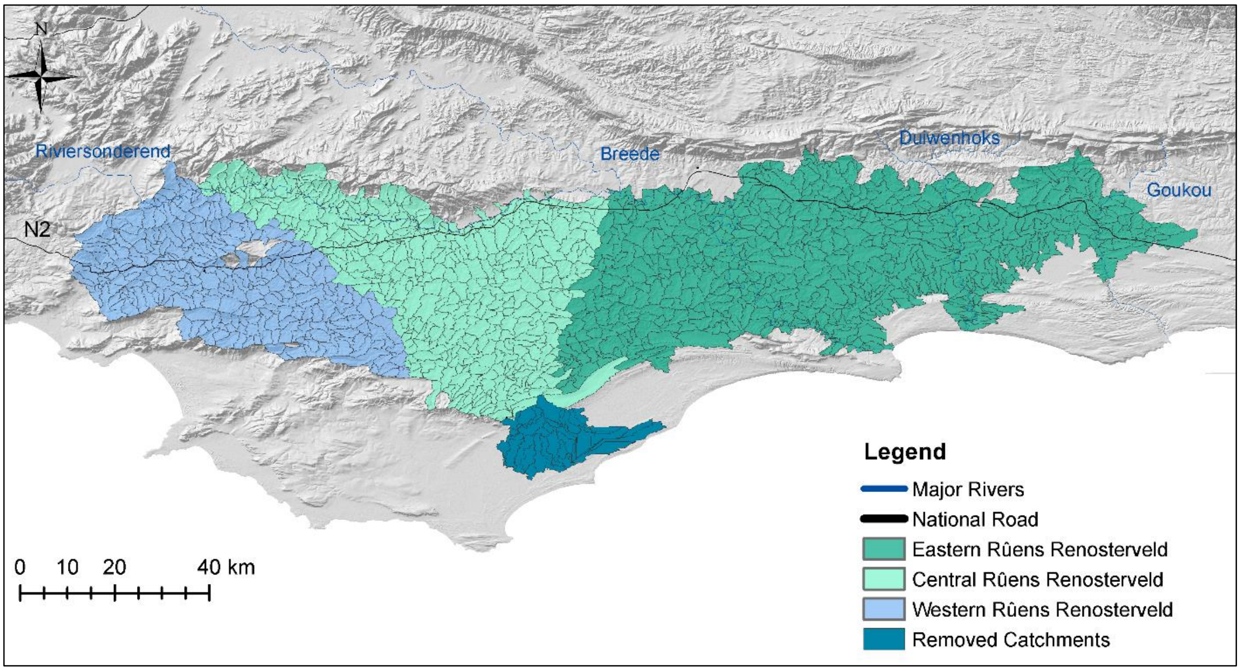


Figure 3.2 Preliminarily delineated catchments with 52 central catchments removed.

Two analyses were then performed. The first analysis did not take contemporary landuse into account, rather giving an approximation of similarity under 'natural', undisturbed vegetative conditions (landuse was not included as an explanatory variable in the cluster analysis). This assumes relatively homogenous vegetation across the landscape for each renosterveld type. The second analysis included the nine landuse categories as variables and aimed to depict contemporary catchment similarity.

In total, 1600 catchments were delineated across the region – 329 in the Western (blue polygons in Figure 3.2), 528 in the Central (light green polygons) and 743 in Eastern Rûens Shale Renosterveld (dark green polygons). The central region's vegetative range extends into the coastal plain to the south (see Figure 3.2), however the topography is extremely flat resulting in catchment boundaries (watersheds) being poorly approximated due to a lack of a nuanced flow direction model. This resulted in 52 catchments in the central region being removed from the data set. Furthermore, some insignificantly small catchments (some <0.01 km^2) were delineated. All catchments with an area less than one standard deviation for each vegetative area were subsequently excluded from analysis. Thus, 248 (Western), 418 (Central) and 651 (Eastern) final catchments were analysed.

3.3 RESULTS

Table 3.2 reports the level of transformation the region has undergone since anthropogenic activities have extended into the area. There is a predominant shift from renosterveld vegetation to agricultural landuses with 56%, 70% and 67% of the land having been converted to cultivated fields in the Western, Central and Eastern Rûens Shale Renosterveld respectively. There are also some thicket, woodlands and forest communities within the broader renosterveld landscape (shrubs and grasses) and surrounding buffer zone. Much of the remaining indigenous vegetation is highly fragmented and prone to invasive alien plants such as pines and black wattle, particularly in riparian zones.

Table 3.2 Landuse types in the Overberg region with the percentage degree of agricultural transformation of the Rûens Shale Renosterveld landscape and surrounding areas.

Landuse (% of total area)	Western Rûens Renosterveld	Central Rûens Renosterveld	Eastern Rûens Renosterveld
Bare ground area (%)	1.99	0.33	0.08
Cultivated area (%)	56.4	70.2	67.2
Forest area (%)	1.07	0.67	0.40
Other area (%)	0.02	0.03	0.02
Shrub and grass area (%)	34.3	22.2	23.5
Thicket and woodland area (%)	4.69	4.67	6.39
Urban area (%)	0.29	0.46	0.20
Water area (%)	0.13	0.04	0.62
Wetland area (%)	1.13	1.36	1.56

Given this substantial degree of degradation and fragmentation it was deemed necessary to address catchment physiographic similarity both in an untransformed landscape (assuming a homogenously vegetated landscape) and under current (transformed) conditions. The following sections outline the catchment similarity for delineated quinary catchments under the untransformed (Section 3.3.1) and transformed (Section 3.3.2) scenarios.

3.3.1 Catchment physiographic similarity in an untransformed landscape

Figure 3.3 displays the spatial distribution of catchment clusters for the three shale renosterveld landscapes under natural, homogenous vegetation conditions (landuse not included as a variable for analysis). Under these conditions it is assumed that the vegetation in each region is relatively similar in terms of structure. Vegetative structure can affect primary and secondary interception rates of rainfalls, thus affecting the runoff in each catchment. There certainly are variations in structure at community levels, for instance where the natural shrub and grassland of renosterveld vegetation grades into thicket communities closer to water courses, although this signal is most likely less important at a landscape level in the renosterveld (a grassy shrubland).

However, the influence of agricultural transformation is not considered under these conditions and the results represent an approximation of catchment physiographic similarity prior to widespread interference and disturbance by human activities.

In Figure 3.3A the Western Rûens Shale Renosterveld shows a high degree of similarity towards the interior of the region (Cluster 3 – darker blue) while the boundary catchments also display clustering (Cluster 4 – dark purple). This is unsurprising as the boundary conditions of the area represent a transition into more mountainous topography, while soil characteristics (clay percentage and depth) also change as evidenced by the shift in vegetation to fynbos (see Figure 3.1). The importance of the variables (ranks 1 to 12) in the analysis (Table 3.3) indicates that elevation range is the primary predictor of catchment clusters, while area and soil factors rank second in importance. A strong west-east rainfall gradient exists as the western boundary catchments show mean annual rainfalls of 880 mm/a, dropping off towards the interior to <480 mm/a. Maximum elevations are also greater in the boundary catchments. There is a marked change in clay percentage in the soils (Figure 3.4B) that varies from approximately 3% in the western boundary catchments to >17% in the interior. The interior also shows little variation in terms of average soil depth (Figure 3.4A).

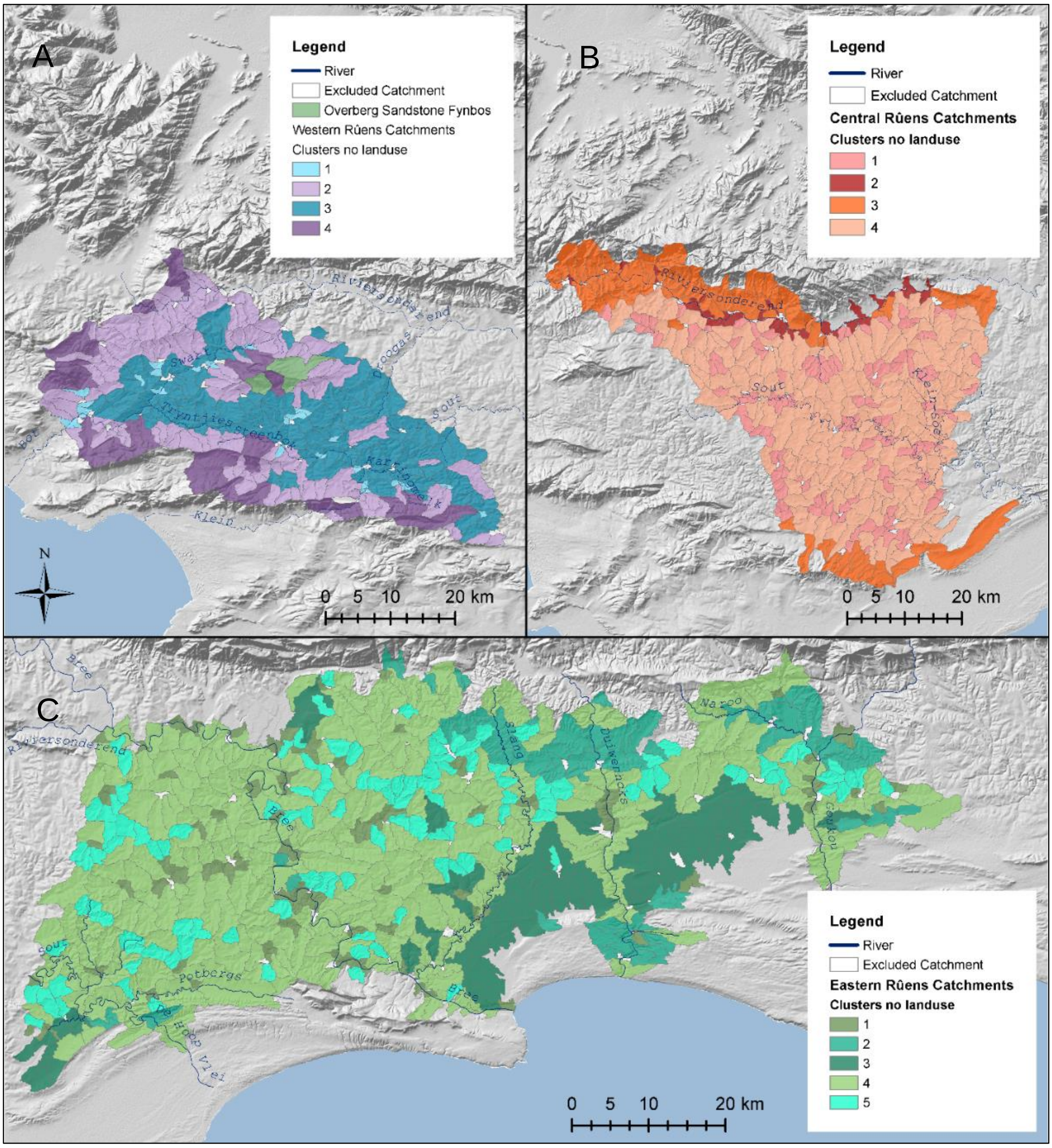


Figure 3.3 Physiographically similar subcatchments under renosterveld vegetation based on cluster analysis of the Overberg region's (A) Western, (B) Central and (C) Eastern Rûens Shale Renosterveld catchments.

In the central region (Table 3.3) elevation range is again the most important variable affecting cluster selection. Soil type (average clay percentage), average slope and catchment area are also important variables. Once again, strong clustering towards the perimeter of the central region exists (Figure 3.3B – Clusters 2 and 3).

Table 3.3 Importance of variables (classification and regression tree (C&RT) relative variable importance) determining the Rûens Shale Renosterveld catchment clusters in an untransformed landscape.

Rank	Western Rûens Shale Clusters		Central Rûens Shale Clusters		Eastern Rûens Shale Clusters	
1	Elevation range (m)	1.00	Elevation range (m)	1.00	Mean slope (degrees)	1.00
2	Catchment area (km^2)	0.79	Mean clay (%)	0.89	Catchment area (km^2)	0.90
3	Mean clay (%)	0.71	Mean slope (degrees)	0.84	Elevation range (m)	0.82
4	Perimeter (km)	0.65	Catchment area (km^2)	0.67	Mean soil depth (mm)	0.69
5	Mean slope (degrees)	0.53	Maximum flow accumulation	0.66	Perimeter (km)	0.67
6	Mean rainfall (mm/a)	0.49	Mean soil depth (mm)	0.65	Stream length	0.64
7	Mean soil depth (mm)	0.40	Shape parameter	0.52	Mean clay (%)	0.53
8	Land type count	0.38	Mean rainfall (mm/a)	0.48	Maximum flow accumulation	0.49
9	Stream length	0.29	Perimeter (km)	0.46	Shape parameter	0.48
10	Mean slope curvature	0.28	Land type count	0.46	Mean rainfall (mm/a)	0.48
11	Maximum flow accumulation	0.24	Stream length	0.33	Land type count	0.15
12	Shape parameter	0.07	Mean slope curvature	0.33	Mean slope curvature	0.12

Quinary catchments adjacent to major rivers are also characterised by larger maximum flow accumulations (>52 000 000 pixels) than in the other catchments in the area, thus marking them as a distinct cluster. Most of the interior of the central region consists of undulating topography with mean catchment slopes displaying a north-south gradient where slope angles of >17° exist on the northern boundary catchments in the foothills of the Riviersonderend Mountains, dropping to 2-3° for most of the interior. There is very little variation in mean annual precipitation across the central region (400-470 mm/a). Average clay percentages in the central catchments show little variation, with values ranging from 15 to 17%, whereas the perimeter catchments have much lower clay content values (3-7%).

The spatial pattern of catchment clusters in the Eastern Rûens shows a greater randomness than the patterns exhibited by the Western and Central Rûens regions. The variables accounting for this clustering pattern are the average slope, area and elevation range of these catchments, with the soil factors of soil depth and clay percentage also being important.

Some clustering exists in the landscape along river lines, but these clusters are poorly defined. There is a notable cluster (dark green) toward the south of the Slang and Duiwenhoks Rivers. This is attributable to the flatness of the area where the mean slope of these catchments is approximately 1°.

Interestingly, a small cluster of catchments follows the path of the Riviersonderend (Cluster 2) – a major river in the Western Cape – where soils (Figure 3.4A and B) and topographical features are substantially different from those of the surrounding area.

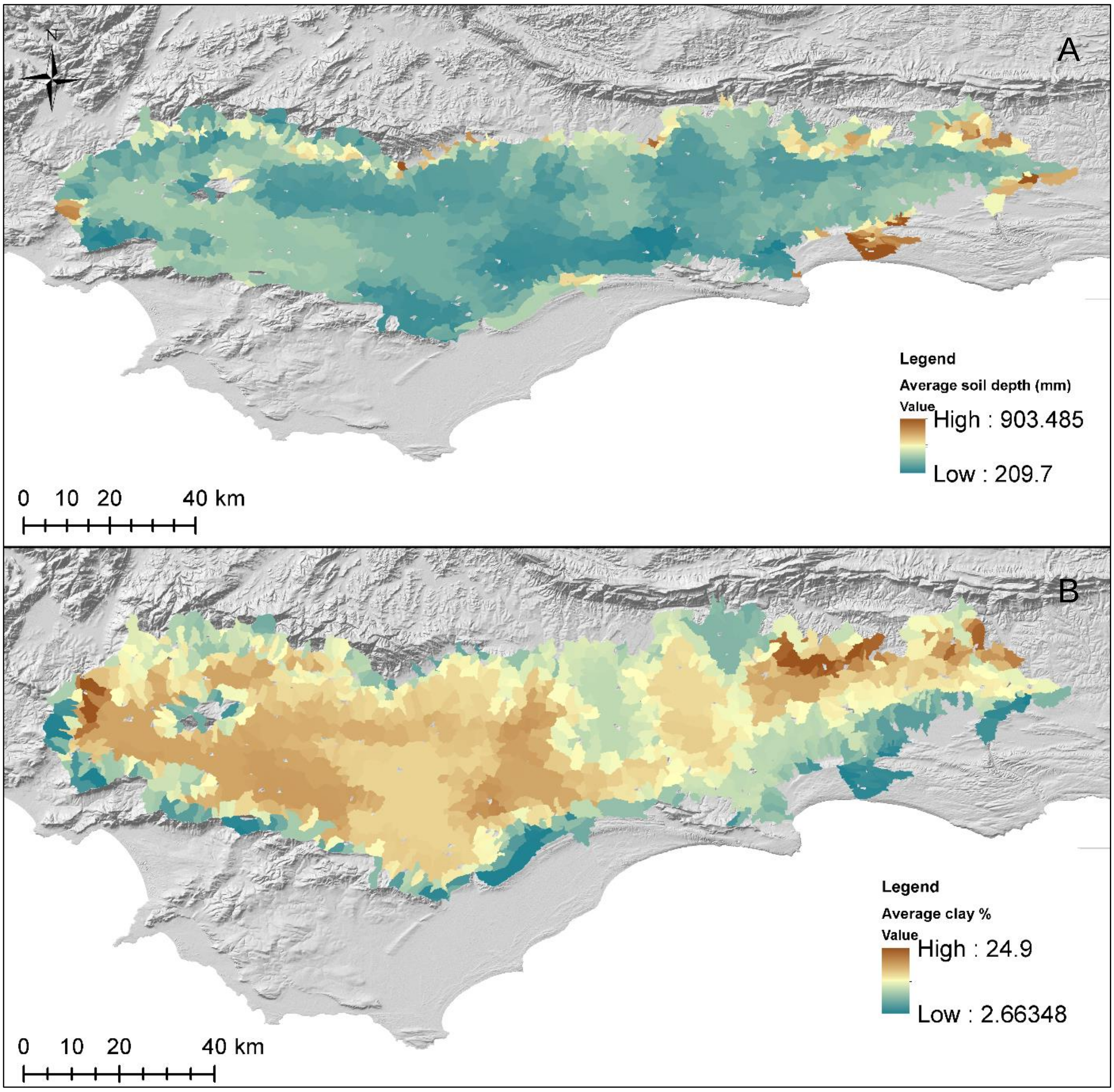


Figure 3.4 Mean soil depth (A) and clay percentage in surface soils (B) for delineated catchments in the Overberg region.

Figure 3.4 reveals that soils in the Overberg are relatively shallow, most ranging between 200 and 300 mm but also high in clay content, where most are >20%. Deeper soils are largely linked to rivers (Figure 3.4A), while lower clay content soils are located in the fringe catchments on the outskirts of the Overberg (Figure 3.4B).

3.3.2 Catchment physiographic similarity within a transformed landscape

When accounting for contemporary landuse, the physiographic similarity of delineated catchments in the Overberg shows some degree of change, particularly in the Eastern Rûens Shale Renosterveld (see Figure 3.5C). In the western region strong clustering still exists in the interior of the area (Clusters 1 and 2) where soils, topographic factors and rainfall show little variation and boundary catchments are clustered together (Clusters 3 and 4). Regarding the importance of the variables in this region, landuse plays a substantial role in cluster allocation with percentage of catchment covered by shrubs and grasses, forest, cultivated land and water emerge as the most important variables accounting for cluster nearness. Topographic and soil factors are also important. In the Western Rûens catchments (Figure 3.5A) boundary catchments show a much higher proportion of shrub and grassland cover than those toward the interior of the region where agricultural landuse dominates the landscape. This spatial distribution conforms well to the topographic and soil features of the region so creating favourable conditions for agriculture in the interior away from the steeper topography and shallower, sandier soils associated with the boundary catchments. There is one catchment in the north of the area that has a much higher share of forest cover (9%) than the majority of the catchments which have no forest cover at all.

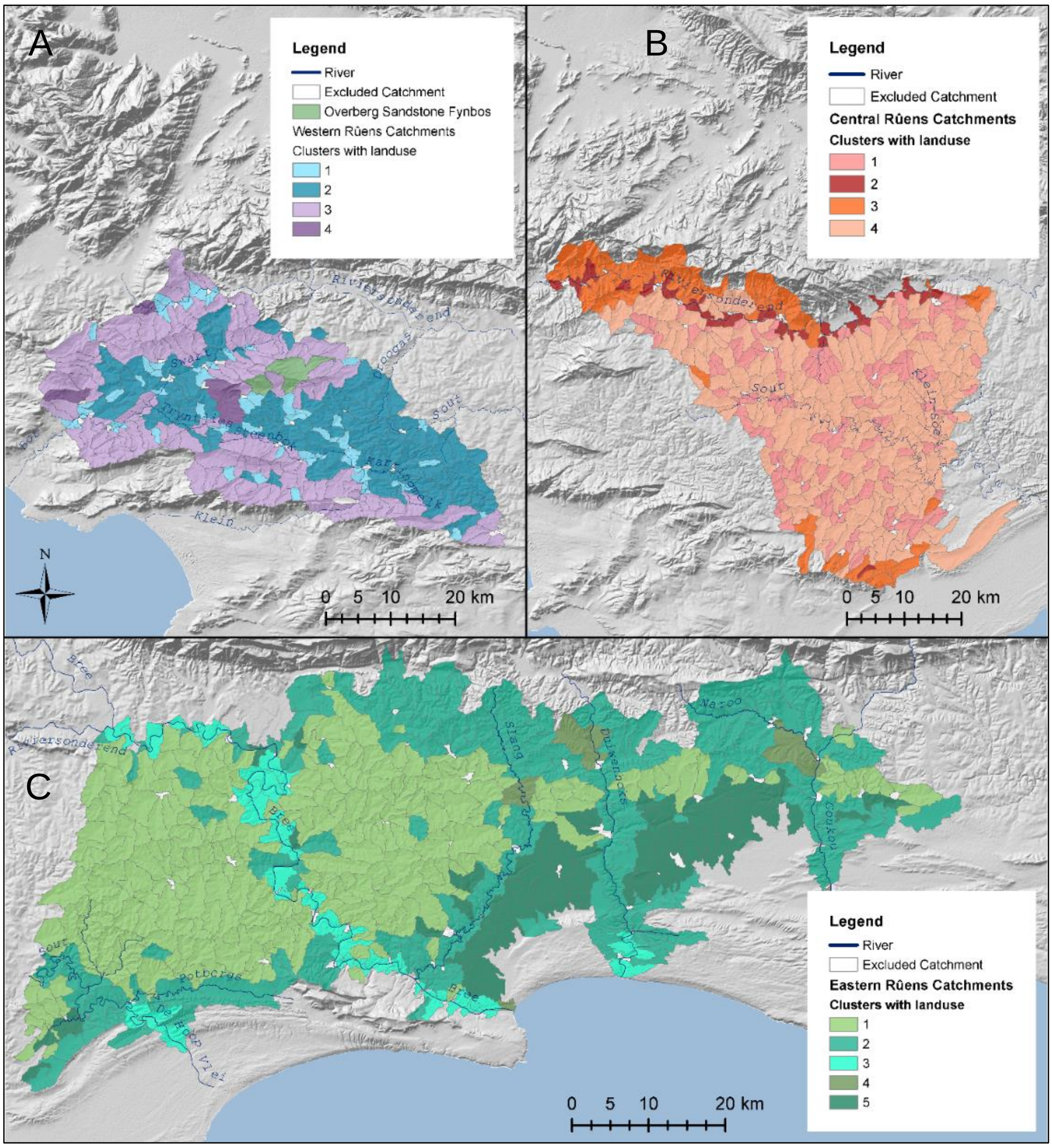


Figure 3.5 Physiographically similar subcatchments under present day conditions – a transformed landscape – based on cluster analysis of the Overberg region's (A) Western, (B) Central and (C) Eastern Rûens Renosterveld catchments.

In the Central Rûens (Figure 3.5B) a similar spatial pattern to the undisturbed clustering emerges. Again, the strong influence of the Riviersonderend can be seen (Cluster 2), while boundary catchments are also clustered together (Cluster 3). Landuse and topographic factors are the most important regarding cluster selection (Table 3.4). It is apparent that the central region has undergone the greatest degree of agricultural transformation within this landscape so that the percentage of catchment covered by agricultural lands is the primary explanatory variable. This is closely followed by mean slope and elevation range (as was the case for an untransformed landscape). Other landuse types, such as shrub and grasses (and thicket vegetation along the northern boundary), as well as soil factors are important, while the effect of maximum flow accumulation is shown strongly by the Riviersonderend flow path.

Table 3.4 Importance of variables (C&RT relative variable importance) determining the Rûens Shale Renosterveld catchment clusters under contemporary landuse.

Rank	Western Rûens Shale Clusters		Central Rûens Shale Clusters		Eastern Rûens Shale Clusters	
1	Landuse shrub & grasses	1.00	Landuse cultivated	1.00	Mean slope (degrees)	1.00
2	Landuse forest	0.86	Mean slope (degrees)	0.94	Mean rainfall (mm/a)	0.86
3	Landuse cultivated	0.85	Elevation range (m)	0.90	Mean clay (%)	0.82
4	Landuse water	0.85	Landuse shrub & grasses	0.88	Mean soil depth (mm)	0.82
5	Maximum flow accumulation	0.84	Landuse thicket & woodland	0.81	Landuse cultivated	0.79
6	Mean clay (%)	0.83	Mean clay (%)	0.76	Landuse shrub & grasses	0.74
7	Catchment area (km^2)	0.82	Maximum flow accumulation	0.68	Landuse wetland	0.73
8	Mean soil depth (mm)	0.78	Mean soil depth (mm)	0.62	Elevation range (m)	0.71
9	Mean slope (degrees)	0.76	Catchment area (km^2)	0.57	Shape parameter	0.68
10	Elevation range (m)	0.76	Mean rainfall (mm/a)	0.56	Landuse thicket & woodland	0.59
11	Landuse wetland	0.74	Perimeter (km)	0.54	Landuse water	0.57
12	Landuse thicket & woodland	0.70	Stream length	0.41	Maximum flow accumulation	0.44
13	Perimeter (km)	0.66	Landuse bare ground	0.37	Landuse bare ground	0.34
14	Stream length	0.60	Landuse urban area	0.32	Perimeter (km)	0.26
15	Land type count	0.58	Shape parameter	0.30	Catchment area (km^2)	0.22
16	Mean rainfall (mm/a)	0.54	Landuse wetland	0.29	Landuse other	0.19
17	Mean slope curvature	0.35	Mean slope curvature	0.27	Mean slope curvature	0.18
18	Landuse urban area	0.29	Land type count	0.26	Land type count	0.17
19	Landuse bare ground	0.25	Landuse water	0.17	Stream length	0.17
20	Shape parameter	0.17	Landuse forest	0.15	Landuse forest	0.12
21	Landuse other	0.03	Landuse other	0.10	Landuse urban area	0.12

The Eastern Rûens manifests the greatest change in terms of cluster distribution and groupings. Here the larger rivers – the Breede (Cluster 3), Slang, Duiwenhoks and Goukou (Cluster 2) –

are much more clearly defined, while a distinct clustering of catchments in the interior between these north to south-flowing rivers is evident (Cluster 1). The spatial pattern of these catchment clusters appears much less random than the pattern exhibited in Figure 3.3C, thereby disclosing the importance of landuse transformation to general catchment physiographic similarity. Cluster 5 (Figure 3.5C), however, remains largely the same as its equivalent in Figure 3.3C (Cluster 3). Once again this shows the importance of topographical characteristics like catchment slope in this analysis. It is noteworthy that the mean rainfall variable is much more important to cluster assignment here than in the previous analysis (no landuse).

The relative cover of wetland, thicket and woodland vegetation, as well as the less-clayey soils, coupled with the water landuse and maximum flow accumulation variables, help define the catchments closer to large rivers in clusters. Soil depth varies greatly with greater depths in the boundary regions (particularly in the north) and in the lower course of the Duiwenhoks River (>900 mm) (Figure 3.2), whereas the interior of the region shows reasonably little variation (290-350 mm). Agriculture is the dominant landuse in the landscape. There exists a marked west-east rainfall gradient with a mean annual precipitation of >540 mm west of the Breede River dropping to 440 mm in the eastern-most catchments and <390 mm in the northern boundary catchments near the Duiwenhoks River.

3.4 DISCUSSION

The results presented here show the marked impact of topographical features on catchment physiographic similarity. Physical characteristics such as soil clay percentage and soil depth also play an important role in this determination. Some noteworthy spatial patterns emerge in all three vegetation types where boundary catchments, which exhibit changes in topography and soils, are clustered together and interior catchments with similar characteristics form distinct clusters. Large rivers are easily identifiable in the results. These results also bring the question into focus of whether hydrologically-similar catchments (homogenous catchments) should be located geographically adjacent to one another or not (Parajka, Merz & Blöschl 2005). It is argued that, in terms of hydrological response, catchments are not homogenous simply because they are in close geographic proximity to one another (Shu & Burn 2003). In fact, in some regionalisation studies donor catchments are assigned based on physiographic similarity (Burn & Boorman 1993). Figures 3.3 and 3.4 show that although there is certainly some degree of spatial coherence to catchment similarity (distance matters!) it is not correct to assume that adjacent catchments are necessarily similar physiographically and would fall into the same cluster.

The degree to which landscape transformation (particularly through the agricultural sector) has an effect on catchment similarity is also evident. This was most noticeable in the Eastern Rûens Shale region where clustering became substantially less random and fits more neatly into geographic zones (catchments in the same cluster located adjacent to one another). Despite the inclusion of a variety of landuses in this analysis, topographic features – particularly mean slope and elevation range – still exhibit a great influence on cluster selection. This further illustrates how important the physical characteristics of terrain are. They are crucial elements of overland flow generation (Mashimbye, De Clercq & Van Niekerk 2014) and determinants of catchment similarity in terms of physiography and hydrological response. Thus the use and choice of DEMs to delineate catchments and calculate these slope elements is vital for nuanced studies of smaller catchments. High-resolution terrain and surface models should be employed for these purposes. High-resolution DEMs permit high-accuracy, non-biased delineation of watershed lines, appropriate for high-resolution, fine-scale hydrological surface flow modelling. The use of GIS software in the exercise also greatly shortens the processing time required for such detailed delineations and analyses.

In the Rûens Shale Renosterveld landscape it is clear that catchments towards the interior of these vegetative regions behave similarly and are by and large clustered together. Fringe (perimeter) catchments also exhibit similar cluster groupings, while those quinary catchments adjacent to large rivers such as the Riviersonderend, Breë, Duiwenhoks and Goukou display a high degree of clustering.

A further application of this research is the identification of paired catchments for paired catchment analyses in ungauged areas. As agglomerative clustering methods begin by assigning each catchment to its own cluster and then finding the nearest cluster (catchment), it is possible to identify two individual (or a group of) catchments that are most physiographically similar to one another.

3.5 CONCLUSION

A similarity approach to catchment regionalisation (Nathan & McMahon 1990; Parajka, Merz & Blöschl 2005) was adapted in this chapter by first delineating quinary catchments and then determining their similarity based on a range of physical parameters. This signals the approach's potential use in ungauged areas or regions with low gauging network densities. The use of GIS software is of great benefit in first delineating small catchments for quinary level analysis. GIS data sets can then be used to interpret catchment physiographic similarity in the absence of hydrological information. Homogenous catchments could potentially observe a similar hydrological response to rainfall based on their physiographic features, although this could be tested with the use of regression equations for gauged rivers and streams. Topographic features such as average slope and elevation evidently have particular importance in the determination of physiographic similarity. Landuse change also markedly impacts the overall pattern of catchment similarity in the Overberg region where landuse change has resulted in heterogeneity between catchments where homogeneity previously existed. The results presented here suggest that catchment physiography has been greatly altered by agricultural landuse in the Overberg region. There is a need to evaluate the magnitude of these impacts with regards to streamflows and sediment erosion in Overberg catchment. This is considered in Chapter 4.

CHAPTER 4: ESTIMATING CATCHMENT SEDIMENT YIELD AND THE IMPACTS OF CLIMATIC AND LANDUSE CHANGE IN A DATA-SCARCE SETTING: A HYDROLOGICAL MODEL FOR THE BOT RIVER, SOUTH AFRICA

4.1 INTRODUCTION

Hydrological processes (both surface and groundwater), sediment delivery and pollution of waterbodies are greatly affected by climatic variability (Liu et al. 2015; Zhao et al. 2010) and by the transformation of natural landscapes to agricultural fields (Evans et al. 2019; Gleeson & Richter 2018; Sharpley et al. 2015). Altered flow regimes and sediment dynamics are driven by changes in precipitation dynamics (volume, intensity and frequency) which affect the conversion of precipitation to direct surface runoff or infiltration into soils, so affecting the flow components of a stream (Chorley 1978; Ward 1967) and soil erosion on catchment hillslopes (Burt et al. 2016). Furthermore, surface runoff and soil erosion are often associated with agricultural practices (Duvert et al. 2010; Nu-Fang et al. 2011). Erosion rates from ploughed fields are typically one to two orders of magnitude higher than erosion under natural vegetation conditions, eventually resulting in declining field productivity (Montgomery 2007). Increases in rates of soil erosion and delivery of sediment to waterbodies are also considered to be significant environmental concerns for water resource managers (Collins & Zhang 2016; Duvert et al. 2010; Ongley 1996). As a consequence of climatic change and the expansion and intensified use of croplands and pasture, the delivery of sediment (particularly fine-grained sediment) to rivers can change the physico-chemical and biological conditions of these habitats and affect ecosystem health (Collins & Zhang 2016; Ryan 1991).

Soil erosion processes are complex and vary considerably across diverse spatial and temporal scales (Gwapedza et al. 2021). Sediment delivery in catchments is mainly driven by three processes, namely 1) erosion due to raindrop energy and surface runoff causing sheet, rill and inter-rill erosion; 2) mass movements from unstable hillslopes; and 3) bank and riverbed erosion (Fink et al. 2017). Accelerated rates of soil transport have become a major global concern (Duvert et al. 2010; Gwapedza et al. 2021; Montgomery 2007). This trend is alarmingly evident in South Africa where some 25% of all reservoirs in the country have reportedly lost just under one third of their storage capacity due to sedimentation (Msadala et al. 2010). Several local studies have estimated sediment loss in South African catchments based primarily on available data from surveyed reservoirs or rivers (Grenfell & Ellery 2009; Msadala & Basson 2017; Msadala et al. 2010; Rooseboom 1975; Rooseboom et al. 1992) all illustrating the substantial effects of sedimentation in dams and rivers. Despite this, there is little information on the sediment delivery of unmonitored and data-poor catchments in the country. Sediment delivery to streams and rivers affects reservoir capacity through sedimentation as well as the physico-chemical properties of the water itself (De Jonge et al. 2004; Schoumans et al. 2014). In particular, phosphorus delivery to rivers is affected by soil erosion as phosphorus strongly sorbs to mobile colloidal particles (De Jonge et al. 2004).

Assessments of the potential sources of sediment to rivers can greatly assist river management. However, many South African rivers are poorly-studied and data for these catchments are sparse – particularly relating to sediment contributions from hillslopes to rivers. As such, it is often a challenging and expensive task to achieve representative observations of sediment erosion in rivers (Gwapedza et al. 2021). However, key hydrological processes that consider the interconnections between surface and subsurface flows as well as catchment erosion can be simulated using rainfall–runoff models (Deb, Kiem & Willgoose 2019; Fink et al. 2017; Li et al. 2016; Watson et al. 2018). These hydrological models can provide estimates of the various flow components and simulate surface runoff, streamflows and sediment dynamics although they are often limited by the lack of available data.

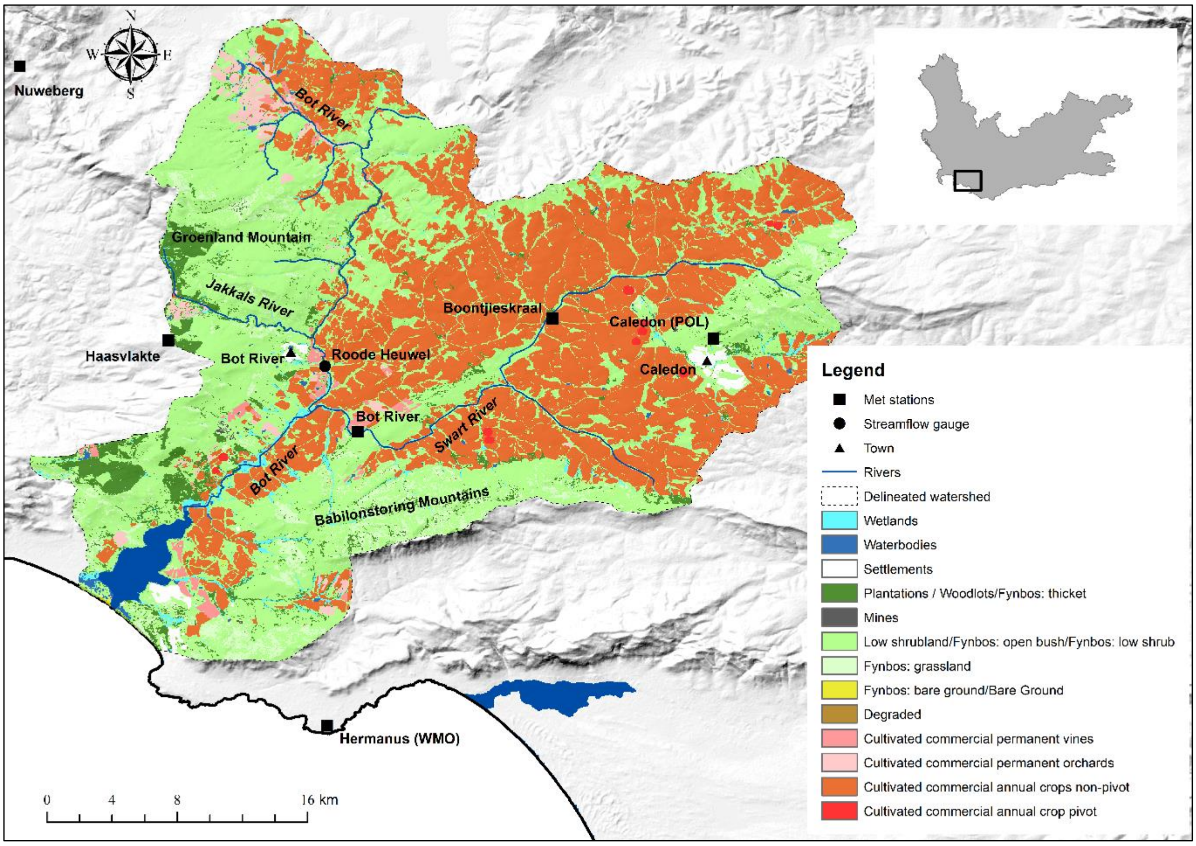
Rainfall–runoff models are often categorised as lumped, semi-distributed or fully distributed versions (Deb & Kiem 2020; Watson, Midgley, et al. 2021). Lumped rainfall–runoff models function by aggregating runoff-generating variables for an entire catchment to represent an overall bulk catchment response (Beven 2012; Tran, De Niel & Willems 2018). However, when catchment conditions are such that there is substantial variation in topographical and land surface features within watershed boundaries, the nuanced spatial variations in recharge mechanisms are ignored (Watson, Kralisch, et al. 2021). The fully distributed and semi-distributed model approach (e.g. JAMS/J2000; Krause, 2001) uses hydrological response units (HRUs) within catchments as the spatial units representing homogeneity in topography and runoff-generating factors (Watson et al. 2019). As such, the flow components of direct surface runoff, soil interflow and baseflow for each river reach and the basin can be estimated. Therefore, distributed models can be used to simulate catchment hydrology and sediment dynamics even where only limited data exist.

Erosion modelling in response to variations in landuse and hydrological processes (flow components) is crucial to managing rivers and their catchments. Climate analysis and projections for South Africa suggest that extreme rainfall events have become more intense and this will likely increase in the future (De Waal, Chapman & Kemp 2017). These events affect the flow components of catchments as direct surface runoff increases under higher rainfall intensity. However, data scarcity presents formidable challenges to developing these models. This study considers the assessment of sediment delivery in response to alterations in the flow components and changing landuse in the Bot River catchment, Western Cape, South Africa (Figure 4.1). The purpose is to explore the capabilities of the JAMS/J2000 software to model erosion in response to alterations to anticipated climate change and landuse inputs using several landuse and flow component scenarios.

The Bot River is a rare example of a closed estuary system with the result that detailed investigations of the Bot River lagoon have been undertaken (Bally, McQuaid & Pierce 1985; Bennett 1985; Branch et al. 1985; Koop, Bally & McQuaid 1983; Heÿl & Currie 1985; Van Heerden 1985). However, despite these efforts and a large research project conducted on the estuary in the 1980s (Sloan, Branch & Bally 1985) that mainly considered the effects on estuarine dynamics and habitats of artificially opening the mouth, little research has considered the entire Bot River catchment and its sediment dynamics. The catchment is relatively data-poor, with only one streamflow gauge and limited meteorological data. Despite this paucity, a combination of remotely sensed data inputs (digital surface model and landuse data sets) and observed and modelled meteorological inputs have enabled the application of a distributed JAMS/J2000 rainfall–runoff model to the catchment. This model was used to approximate the relative flow component contributions and to simulate sediment delivery for the Bot River under three landuse scenarios. Thus, particular attention is given to the effect of landuse change on modelled sediment yield in the catchment and the implications for nutrient loading and sedimentation of reservoirs in data-poor regions.

4.2 STUDY SITE

The Bot River catchment (Figure 4.1) encompasses an area of 900-1000 km^2 (Bally 1987; Van Niekerk, Van der Merwe & Huizinga 2005). Major tributaries of the Bot River are the Swart and Jakkals Rivers flowing in from the east and west respectively. The average annual discharge measured at the Department of Water and Sanitation (DWS) Roode Heuwel gauge (G4H014-A01) near the town of Bot River is 22.6 x 10^6 m^3/a (Department of Water and Sanitation 2021). Mean surface runoff has been assessed to be between 47 x 106 m^3/a and 116 x 10^6 m^3/a (Jezewski & Roberts 1986; Noble & Hemens 1978; Van Niekerk, Van der Merwe & Huizinga 2005).



Data source: Department of Forestry, Fisheries and the Environment (2018)

Figure 4.1 Bot River catchment with landuse classes based on South African national land cover 2018 data (Department of Forestry, Fisheries and the Environment 2018), meteorological station locations and the stream gauge used for the JAMS/J2000 model.

The catchment falls in a winter rainfall zone with mean annual precipitation on the Groenland Mountain of 1100 mm/a (1000 m altitude), reducing to 450 mm/a at lower elevations towards the centre and east of the catchment (0-200 m altitude) (Bailey & Pitman 2015). Most of the flow contribution to the Bot River estuary stems from the Groenland Mountain and the Jakkals River. The Swart River to the east of the catchment is conspicuously dry in the summer months (although it does contribute strong flow in the winter). As sediment delivery is linked to sediment availability (Grenfell & Ellery 2009), contributions from hillslopes throughout the catchment are likely to peak during the first flush of the winter season when cultivated fields are still bare but rainfall enables downslope movement of sediment into watercourses. Sediment delivery is lower during the summer, unless initiated by a strong storm event.

The catchment geology is dominated by Bokkeveld Group shales in low-lying areas and Table Mountain Group (TMG) sandstone at higher elevations. The catchment comprises lithic, duplex, plinthic, gleyic and cumulic soils within the various landtypes (Fey 2010). Lithic soils in the Bot River catchment are characteristically shallow and stony and erode to form convex slope crests and gentler footslopes (Fey 2010). The presence of duplex soils in the area indicates the enrichment of the subsoil with clay. There is thus a marked increase in clay composition in the B horizon when compared with the horizon above it (Fey 2010). The shale-derived, duplex clay soils in the catchment have been converted to agricultural use and contribute relatively high inputs of silt and other erosion products from hillslopes (Koop, Bally & McQuaid 1983).

Rûens Shale Renosterveld is the principal natural vegetation in the area on nutrient-rich, shale-derived soils, while the nutrient-poor slopes of the mountainous areas are mainly covered by mountain fynbos (Curtis & Bond, 2013; Rebelo et al., 2006). However, much of the land area once covered by renosterveld has been converted to cereal and oilseed agriculture (Curtis 2016; Curtis & Bond 2013). The riparian zone of the Bot River is overrun by a melange of invasive tree and plant species with extensive reed beds of *Phragmites* and *Scirpus* occurring south of the Bot and Swart Rivers' confluence where the catchment gradient lowers substantially (Koop, Bally & McQuaid 1983).

4.3 METHODS

Descriptions of the input data (Section 4.3.1), a regionalization of climate input (Section 4.3.2), calibration and validation of the simulated flows (Section 4.3.3) and the estimated sediment delivery using the MUSLE component (Section 4.3.4) are given in this section. The JAMS/J2000 (Krause 2001) process-orientated ecohydrological model which simulates small-scale and hillslope processes was applied to the Bot River catchment. The JAMS/J2000 model required distributed climate, soil, landuse and geological data to delineate HRUs and a reach network[17]. A further model component, *MUSLE*, was added to the modular component list to simulate sediment delivery in the river system. To evaluate how landuse change affects sediment delivery, the streamflow and erosion model was implemented and validated using a number of scenarios (Table 4.1).

Table 4.1 Various streamflow and sediment yield scenarios used in Bot River analysis.

Streamflow Scenario	Conditions	Sediment Yield Scenario	Conditions
A	1990 landuse	X	1990 landuse
B	1990 landuse, relaxed model parameters		
C	2018 landuse	Y	2018 landuse
		Z	Undisturbed, "natural" conditions

[17] A river reach refers to a spatial unit with similar physiographic and hydrological characteristics

Scenarios included the 1990 landuse data set (Scenarios A, B and X), the 2018 landuse data set (Scenario C and Y) as well as an undisturbed, natural landcover dataset (Scenario Z). To understand the impact of climatic variability on streamflows and erosion, the model was run for two flow component scenarios – high surface runoff and low surface runoff.

4.3.1 Model input data

The model inputs for JAMS/J2000 are described below. First meteorological input data are considered (Section 4.3.1.1). This is followed by a description of the landuse data used for the study (Section 4.3.1.2), the soil data input (Section 4.3.1.3), geology data (Section 4.3.1.4) and the data required for delineating HRUs (Section 4.3.1.5).

4.3.1.1 Meteorological and streamflow data

Meteorological data were sourced from the South African Weather Services (SAWS) and Agricultural Research Council (ARC) databases. Meteorological variables included daily rainfall, windspeed, relative humidity, solar radiation and air temperature collected from several stations set up to record only rainfall or as fully automated weather stations (AWSs) (Table 4.2). These data (excluding rainfall) were used as input into the Penman-Monteith equation (Allen et al., 1998) for estimating daily reference evaporation.

Table 4.2 Available meteorological stations and their data for input into the Bot River JAMS/J2000 model.

Station name	Rainfall	Tmin	Tmax	Tmean	Wind speed	Solar radiation	Relative humidity***	Source
Bot River	✓	✓	✓	✓	✓	✓	✓	ARC
Boontjieskraal	✓	×	×	×	×	×	×	SAWS
Caledon POL	✓	×	×	×	×	×	×	SAWS
Cape Town*	×	×	×	×	✓	×	×	SAWS
Elgin experimental farm	✓	×	×	×	×	×	×	SAWS
Grabouw	✓	✓	✓	×	✓	×	×	SAWS
Haasvlakte	✓	×	×	×	×	×	×	SAWS
Hermanus	×	✓	✓	✓	×	×	✓	SAWS
La Motte*	×	×	×	×	×	✓	×	ARC
Nuweberg	✓	×	×	×	×	×	×	SAWS
Paarl*	×	×	×	×	✓	×	×	WMO
Solar radiation data (SODA)**	×	×	×	×	×	✓	×	SODA
Strand*	×	×	×	×	✓	×	×	SAWS
Worcester	×	✓	✓	✓	×	×	✓	SAWS

*Stations used when no data were available in the catchment – regionaliser uses next nearest station.

**Modelled data used in absence of solar radiation information (SODA 2021).

***Calculation made from dew-point temperature.

In total, fourteen stations were used as input data for the Bot River model. Where data records were incomplete the model regionaliser used data from the nearest available stations, thus ensuring a complete data record.

Observed streamflow data for the Bot River were obtained from the only streamflow gauge on the river, the DWS Roode Heuwel gauge (G4H014-A01) near the town of Bot River (Figure 4.1).

4.3.1.2 Landuse input

Input landuse data for the hydrological model were sourced from the 1990 as well as 2018 South African National Land Cover (SANLC) data sets developed by GeoTerraImage SA Pty Ltd for the Department of Forestry, Fisheries and the Environment (DFFE) (Department of Forestry, Fisheries and the Environment 2018). The SANLC 2018 data set (derived from Sentinel-2 satellite imagery) was produced to deliver as close as possible an update of the SANLC 1990 product (which used 30-m resolution Landsat 4 and 5 imagery) in terms of data content, format and landscape representations. The 1990 land cover product has 35 classes and 17 simplified classes, although within the Bot River catchment boundaries only 16 classes are present. The 2018 data set has 73 classes of which 49 are present within the catchment boundaries. To ensure comparability of results, these 49 classes were reclassified using ArcMap 10.6 to align with the 1990 class output. For instance, Artificial dams (2018) were reclassified as Water (1990) because the 1990 classification does not differentiate between different types of waterbodies (refer to Appendix A, Table A1 for a detailed description of reclassified classes). Each landuse class was allocated an albedo, root depth and seal grade (% impervious surfaces) value derived from the literature and previous models in the area (Steudel et al. 2015; Watson et al. 2018; Watson et al. 2019; Watson et al. 2020). Leaf area index (LAI) and vegetation height vary by growing season, particularly in agricultural areas that often lie fallow (in the summer months of November-April for the Bot River area). For this reason, different LAI values were assigned for each growing season (quarterly).

4.3.1.3 Soil data

The Harmonized World Soil Database (HWSD) version 1.2 (HWSD) (Batjes et al. 2012) was used to estimate soil type data – water storage capacity, average soil depth, topsoil (T) and subsoil (S) depth, texture and granulometry. First, unique soil types in the Bot River catchment were identified using their MU_GLOBAL codes. A weighted mean for sand, silt and clay in the topsoil (0-30 cm) and subsoil (30-100 cm) for each unique MU_GLOBAL code was calculated. These weighted mean values were used as input to the Rosetta lite (Schaap, Leij & Van Genuchten 2001) in the HYDRUS 1-D model (Šimůnek, Van Genuchten & Šejna 2006; 2008) to predict available water-holding capacity (AWC) at 0, 59~60 and 1500 kPa (last reported as residual water). Soils were grouped according to their respective water-holding capacities given by medium pore storage (MPS) (0.2-50 µm) and large pore storage (LPS) (>50 µm). Thus, MPS and LPS represent two soil storages that vary in their pore size. LPS and MPS were calculated and then multiplied with the full depth of the soil to obtain the total plant-available water and total drainage for each soil profile (Table 4.3).

Table 4.3 Estimated water-holding capacity of MPS and LPS calculated from the HYDRUS 1-D model (Šimůnek, Van Genuchten & Šejna 2008) based on mean soil depth and texture classes in the Harmonized World Soil Database v1.2 (HWSD).

Soil type	Depth interval	Max soil depth (mm)	Sand, silt, clay (%)	Soil-water content at different pressures (mbar)		LPS[3] suction (0-60mbar)	MPS[4] suction (60-15,000mbar)	Water-holding capacity (mm)	
				0	60			LPS	MPS
Leptosols	T[1]	100	43, 29, 28	0.4129	0.3542	0.0587	0.2802	5.87	28.02
							Total	5.87	28.02
Calcisols	T	300	86, 9, 5	0.3804	0.2799	0.1005	0.2289	30.15	68.67
	S[2]	1000	88, 8, 4	0.3802	0.2774	0.1028	0.2268	71.96	158.76
							Total	102.11	227.43
Acrisols	T	300	57, 19, 24	0.3921	0.3203	0.0718	0.2556	21.54	76.68
	S	1000	48, 17, 35	0.4069	0.344	0.0629	0.2666	44.03	186.62
							Total	65.57	263.3
Cambisols	T	300	69, 19, 12	0.4005	0.3345	0.066	0.2652	19.8	79.56
	S	1000	70, 17, 13	0.4034	0.3387	0.0647	0.2663	45.29	186.41
							Total	65.09	265.97
Regosols	T	300	69, 19, 12	0.3844	0.2929	0.0915	0.2426	27.45	72.78
	S	1000	70, 17, 13	0.384	0.2972	0.0868	0.2431	60.76	170.17
							Total	88.21	242.95
Luvisols	T	300	86, 9, 5	0.3895	0.3135	0.076	0.2547	22.8	76.41
	S	1000	88, 8, 4	0.3936	0.323	0.0706	0.2578	49.42	180.46
							Total	72.22	256.87

Notes: [1] Topsoil [2] Subsoil [3] Large pore storage [4] Medium pore storage

The volume of infiltrated water into the soil is empirically calculated by the JAMS/J2000 model, using the maximum soil infiltration rate and the relative soil saturation deficit. Soil saturation deficits were calculated using the relationship between actual MPS and LPS, the maximum MPS to LPS and their water storage capacity.

4.3.1.4 Geology

The JAMS/J2000 model requires maximum storage capacity and storage capacity coefficients (values based on expert knowledge of quantifiable model parameters for the globe, commonly used in all JAMS/J2000 models) as well as maximum aquifer thickness and recession coefficients as input for both the modelled upper (referring to alluvial and quaternary sediments as well as fractured rocks that can deliver water rapidly) and lower (Bokkeveld Group shales) aquifers to establish the groundwater flow components of the model (RG1 for the upper aquifer and RG2 for the lower aquifer), while also considering the connectivity between these aquifers. The hydrogeological parameters such as maximum and minimum infiltration rates for summer and winter for the catchment were taken from literature (e.g. Conrad, Nel & Wentzel 2004) that has informed other hydrological models in areas with similar geological formations and climate (Watson et al. 2018; Watson et al. 2019; Watson et al. 2020). A 1:250 000 geological map (3319 – Worcester), acquired from the South African Council of Geosciences (Gresse 1997), was used to identify and delineate the geological formations present in the Bot River catchment.

4.3.1.5 Defining hydrological response units

HRUs are fundamental to distributed hydrological models (Flügel 1995) and are calculated by the intersection of geographical information system (GIS) layers of soil type, landuse, geology and topography (Sanzana et al. 2013). For this study, the GRASS-HRU web-based tool (Schwartze 2008; Watson et al. 2020) was used to delineate HRUs for the Bot River catchment. HRU delineation integrated the Roode Heuwel DWS streamflow gauge, a digital elevation model (DEM) (Shuttle Radar Topography Mission: SRTM 30 m), as well as maps of landuse (SANLC), geology (Gresse 1997) and soils (HWSD). The HRU delineation was conducted by:

(1) Filling sinks in the DEM using a standard filling approach and developing flow direction and flow accumulation layers for the catchment. This allowed for the delineation of stream channels and sub-basins within the catchment area. In contrast to other local hydrological models (Watson et al. 2019; Watson et al. 2020) which used the SRTM 90 m DEM, the SRTM 30 m DEM was chosen to give a higher level of reach delineation with a sufficient level of detail to represent all stream segments of hydrological importance. The resultant minimum sub-basin size (made up of several HRUs) for this study was 1.35 km^2.

(2) Generating DEM derivatives (slope, aspect and elevation calculations), sub-basins as well as landuse, soil and geology inputs and combining them in an overlay analysis to delineate HRUs. Small "sliver" HRU polygons (<0.09 km^2) were combined with

neighbouring HRUs. Thus, 9686 HRUs were delineated in total (minimum size = 0.09 km^2, maximum size = 1.62 km^2).

(3) Using the flow accumulation raster to produce a slope length output as well as a flow routing network that allowed for downslope connectivity between HRUs or with specific reach segments. HRUs drained either directly to the delineated streams (reach segments) when adjacent to a reach segment or into another HRU further downslope.

(4) Computing HRU characteristics and statistics including area, elevation, slope, mean slope length and aspect.

4.3.2 Regionalisation of meteorological input

Because meteorological data are given as point data (the location of each meteorological station), these were regionalised across the catchment using the inverse distance weighted (IDW) approach as per Watson et al. (2020). Each HRU was assigned a vector giving the weights for all climate stations contributing to its value. A scaling factor was applied to each HRU based on its elevation and proximity relative to nearby meteorological stations. Therefore the regionalisation of these point data allows for a rainfall surface to be developed that enabled the simulation of surface runoff and infiltration in each HRU.

4.3.3 Streamflow model calibration, validation and sensitivity analysis

Model calibration is usually performed to reduce the difference between modelled and observed dependent variables (e.g. streamflow) by adjusting the model parameters to provide the best prediction of outflow when compared to measured data (Gupta et al. 2009; Watson, Kralisch, et al. 2021). In order to calibrate and validate the model the data record was split into three periods: a) model initialisation from 7 March 1995 to 6 March 1998; b) model calibration from 7 March 1998 to 31 December 2002; and c) model validation from 1 January 2003 to 7 March 2008. These periods were selected as data availability and quality declines substantially pre-1995 and a 10-year runtime for the model was used. A semi-automatic calibration method was used by combining automated and manual procedures (e.g. Boyle, Gupta & Sorooshian 2000; Watson et al. 2018; Watson et al. 2020) to include expert understanding and familiarity with local parameters in the catchment, such as seasonal infiltration capacity. A robust optimisation process (Watson et al. 2020) was used to determine ideal parameters for the model. Calibration of the model parameters was conducted via the non-dominated sorting genetic algorithm (NSGA)-II as described by Deb et al. (2002). The NSGA-II algorithm is designed to calculate the best parameter values from Pareto-optimal solutions using a multi-objective search approach (Deb et al., 2002; Watson et al., 2020).

Validation of model performance involved qualitative visualisations and calculations of several hydrological properties, including observed versus modelled flow, soil moisture and catchment water balances. Statistical efficiency of model performance was based on the Nash-Sutcliffe efficiency (NSE) in standard squared form (E2) (Nash 1970). NSE is commonly applied in hydrological studies for the assessment of model performance but it often over-represents peak flows (Krause, Boyle & Bäse 2005). Thus, logE2 was used for better evaluation of low-flow periods as recommended by Watson et al. (2020). Values closer to one for E2 and logE2 represent better models. Additionally, percent bias (Pbias) (Moriasi et al. 2007) was used to quantify overall model performance. Pbias is a measure of the average tendency of modelled values to be larger or smaller than the observed data. Thus, the ideal Pbias value of 0.0 represents a perfect simulation. Where Pbias is positive the model has an overestimation bias and when negative values are reported, the simulation has a tendency to underestimation. Finally, the Kling-Gupta efficiency (KGE) (Gupta et al. 2009) was used to evaluate correlation, bias and the variability between simulated and measured runoff. KGEs range from $-\infty$ to one where values closer to one represent a more accurate model.

The initial model used the 1990 land cover data set as an input for streamflow estimation. This model was constrained by several parameters that affect infiltration rates using the same constraints established by Watson et al. (2018). However, as these maximum infiltration rates are estimated values, these parameter constraints were relaxed for a separate simulation to allow for greater infiltration and storage in soils in order to determine whether model efficiencies were improved by relaxing the model parameters. If so, further work needs to be conducted regionally on the appropriate values for maximum infiltration rates in summer and winter. Thus, the model was run given two scenarios – Scenario A representing the first set of optimal parameters and Scenario B using a different set of relaxed parameters. The model was also re-run using the 2018 land cover data to determine whether any substantial change in model performance is achieved by the use of a more recent data set (Scenario C).

A regional sensitivity analysis (Monte Carlo analysis) (Watson, Kralisch, et al. 2021) was applied to ascertain the importance of each model parameter in the simulation of streamflow in the Bot River. This analysis is incorporated into the JAMS/J2000 model software.

4.3.4 Estimating sediment delivery

Sediment yield was estimated using the modified universal soil loss equation (MUSLE) (Williams 1975) in combination with a dynamic runoff factor calculated in JAMS/J2000 (Pfennig et al. 2009). In essence, MUSLE calculates soil erosion from a rainfall event for each

HRU. In contrast to the universal soil loss equation (USLE) (Wischmeier & Smith 1965; 1978) that uses rainfall as an estimator of erosive energy (Phuong, Shrestha & Chuong 2017), MUSLE uses the degree of runoff to approximate erosion and sediment yield. The USLE, and its offshoots such as MUSLE, have often been used to estimate sediment erosion in South Africa (Gwapedza et al. 2021). MUSLE is particularly popular as it is available in conventional models such as the soil and water assessment tool (SWAT) (Neitsch et al. 2005) which is frequently applied in South African studies.

The MUSLE follows the basic configuration of the USLE, apart from the rainfall factor being replaced by a runoff factor (Fink et al. 2012; Fink et al. 2017; Pfennig et al. 2009). MUSLE is defined as:

$$SY = 11.8\left(Q \times Q_p \times A_{HRU}\right)^{0.56} \times K \times L \times S \times C \times P \times CFRG \qquad \text{Equation 2}$$

Where:

SY = sediment yield (t) L = slope length factor

Q = volume of runoff (mm) S = slope gradient factor

Qp = peak flow rate (m^3/s) C = cover and management factor

AHRU = area of HRU (m^2) P = support practice factor

K = soil erodibility factor CFRG = coarse fragment content factor

K is defined as the average soil lost per unit of R (the rainfall erosivity factor described by the USLE). The soil units described by the HWSD were assigned K values based on the report by Stewart et al. (1975) which details soil erodibility estimates for various soil texture classes. C values range from 0.001 for well-managed woodland to 1.0 for continuously fallow, tilled fields (Stewart et al. 1975). Selected C values for this study were derived from the literature (Affek et al. 2020) and expert field knowledge of local conditions. The P factor represents support practices that prevent erosion in the landscape. Where no support practices are present P = 1.0 (Phuong, Shrestha & Chuong 2017; Stewart et al. 1975). The model is illustrated schematically in Figure 4.2.

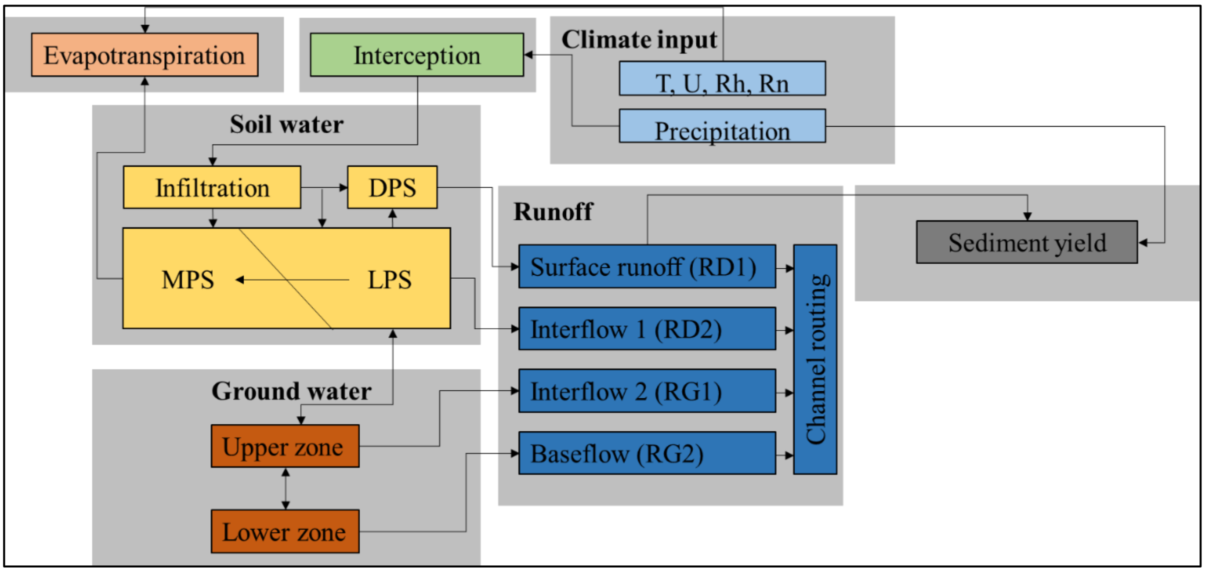


Figure 4.2 Overall model schematic – JAMS/J2000 Bot River model with MUSLE component.

Sediment yield was calculated using both the 1990 SANLC and 2018 SANLC data sets to compare differences between these time periods. A further calculation was performed by converting all agricultural land to natural vegetation to simulate undisturbed (pre-agricultural), natural conditions for comparative purposes. Sediment yield was also estimated using two flow component scenarios (high surface runoff and low surface runoff) to simulate potential sediment responses to changes and variations in climate, particularly rainfall intensity. Scenarios were selected by running 10 000 model iterations and selecting a parameter set for a scenario with a high model efficiency (E2 >0.5) and a high direct surface runoff flow component and another parameter set with high model efficiency and a low direct surface runoff component. Due to a complete paucity of observed sediment delivery data in the catchment, model validation was mainly achieved by comparison of the results with those of other studies done in South Africa, thus giving an estimate of whether the results fall within the same order of magnitude for the area as other studies.

4.3.5 MUSLE limitations

The JAMS/J2000 model with added MUSLE component has several limitations. First, the model does not simulate erosion of sediment from stream banks and bed and so does not include riverbed processes in the estimation of sediment yield. Second, the model assumes continued downstream movement of sediment and does not account for sediment storage in different zones of the river. Third, the impact of fire on soil characteristics (water repellency for instance) is not considered and fourth, the model assumes an infinite supply of sediment from hillslopes.

4.4 RESULTS

The JAMS/J2000 simulation provided runoff and baseflow components as well as an estimate of sediment yield for the catchment. The flow component consists of direct surface runoff (RD1) and interflow (RD2), while the baseflow module comprises contributions from primary (RG1) and secondary (RG2) aquifers. The results of the modelling exercise are presented next by considering the overall model performance, the contributions of the various components and sediment yield under different catchment landuse conditions.

4.4.1 Streamflow model

The streamflow model simulates streamflow which can be compared to observed data. This section outlines the hydrological characteristics of observed flows (Section 4.4.1.1), the performance of simulated streamflows (Section 4.4.1.2) and the sensitivity of each parameter used in the model (Section 4.4.1.3).

4.4.1.1 Observed streamflow

The analysis of observed streamflow data for the Roode Heuwel gauge (G4H014-A01) (Department of Water and Sanitation 2021) shows that there is a marked seasonal variability to streamflow, characterised by strong winter flows and lower summer discharge – a direct response to the seasonal rainfall received in the area. Observed streamflow from April to September constitutes 80% of the annual observed runoff. High inter-annual variability with deviation from the long-term median of annual mean discharge fluctuating between -76% (in 1969) and +170% (2013) exists in the catchment (Figure 4.3). The coefficient of variation (C_v) (Fryirs & Brierley 2013) is 0.54 which represents moderate to high inter-annual flow variability, particularly for a perennial river. Visual inspection of these streamflow deviations shows that much of the modelled period is characterised by lower than normal flows until 2005, after which deviations from the long-term median reveal a positive trend.

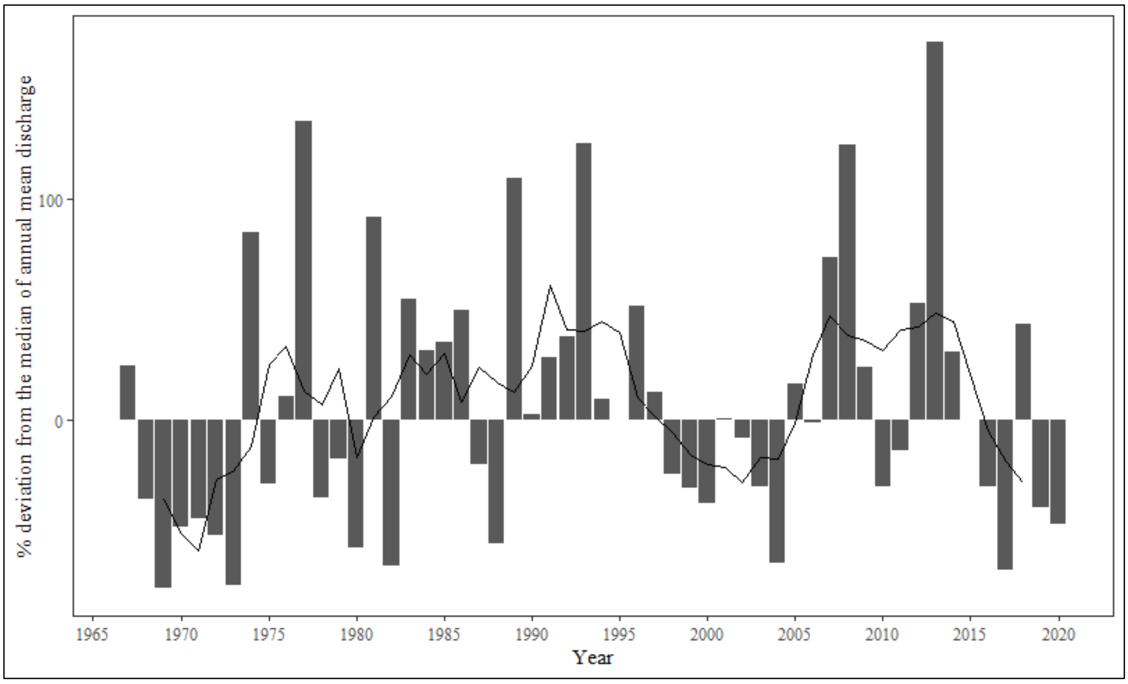


Figure 4.3 The Bot River percentage deviation from the long-term median of annual mean discharge with 5-year running mean represented.

The JAMS/J2000 model uses daily time-step inputs and produces daily outputs. Daily observed and simulated streamflow are illustrated in Figure 4.4. The maximum observed streamflow for the Roode Heuwel gauge during the calibration and validation periods of the model was 70.9 m^3/s on 11 April 2005 and the mean daily streamflow was 0.61 m^3/s. Another extreme event of 43.5 m^3/s occurred on 27 July 2007. Five months of observed data are missing (February to June 2004).

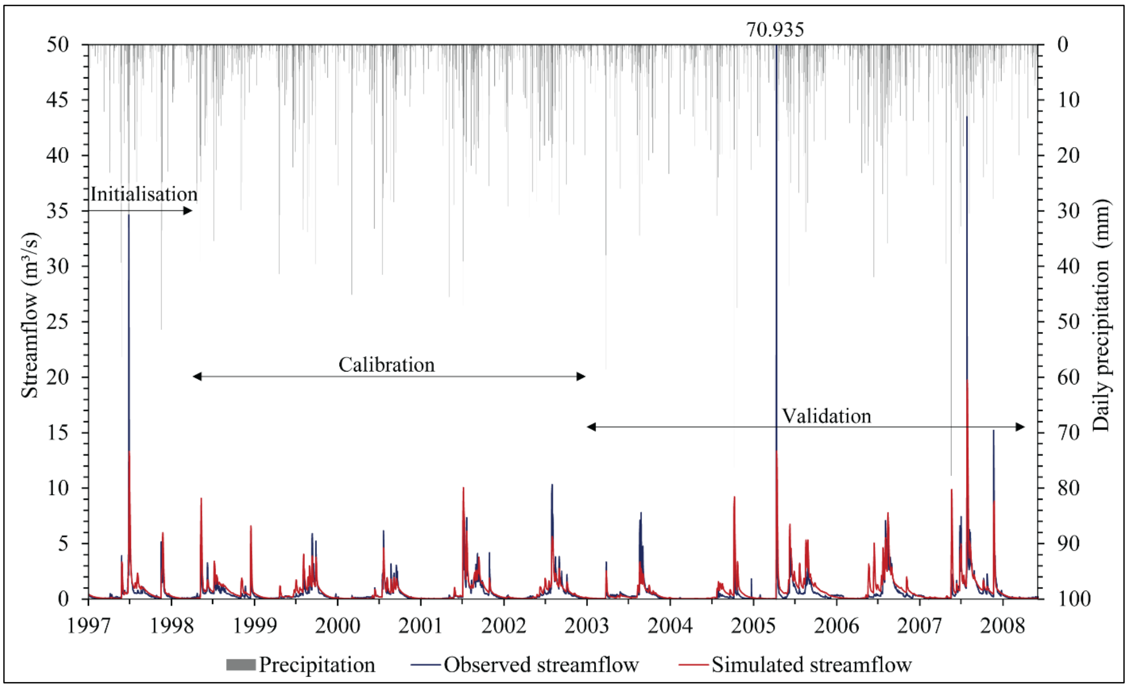


Figure 4.4 Observed and simulated streamflow for Bot River (1997-2008) under 1990 landuse (Scenario A).

There is a marked seasonality to streamflows, with annual maxima occurring during the winter months (May-August) and low flows during dry summers (December-March). Extreme events are more common in the validation period than in the model calibration period.

4.4.1.2 Simulated streamflow

Visual observation and comparison of the observed and simulated streamflow (Figure 4.4) show evidence of strong model performance, especially in the calibration period from 7 March 1998 to 31 December 2002 (R^2 = 0.36). These differences between observed and simulated streamflows are minimal, especially during low flows. This calibration period is characterised by few extreme events (only 19 days exceeded the 98th percentile of mean daily discharge, that is 4.369 m^3/s). During the model validation period (1 January 2003 to 7 March 2008) the model performs relatively poorly. This period is marked by several extreme events (33 exceedances of the 98th percentile) which are all underestimated by the model. However, even during low flows, simulated streamflow does not track the observed record well, often over-estimating streamflow. Figure 4.5 exhibits the monthly averaged observed and simulated streamflows for Scenarios A and B between the model calibration and validation periods. Scenario B (relaxed model parameters) presents a better model fit and produces a better simulation of streamflow than Scenario A. Both models simulate streamflow better in the calibration period, but efficiency drops during model validation. The modelled simulations tend to overestimate flow, particularly in the winter months.

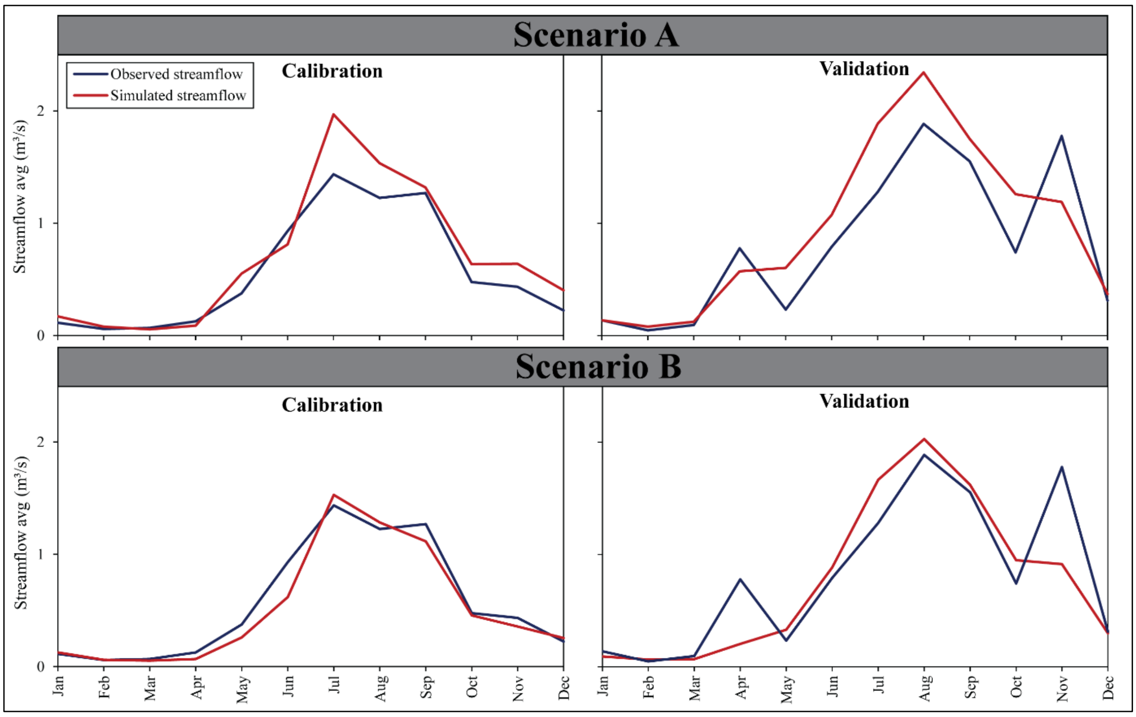


Figure 4.5 Observed and simulated mean monthly streamflow for Scenarios A and B during the model calibration and validation periods.

Considering the lack of recorded meteorological data for the catchment and the extreme climatic conditions of the catchment, the model simulates daily streamflow well. Model performance is evaluated using several assessment criteria as reported in Table 4.4.

Table 4.4 Model evaluation based on the criteria for different simulations.

	Scenario A (optimised model parameters)		Scenario B (relaxed model parameters)		Scenario C (2018 land cover)	
	Calibration 1998-2003	Validation 2003-2008	Calibration 1998-2003	Validation 2003-2008	Calibration 1998-2003	Validation 2003-2008
E1	0.53	0.33	0.68	0.5	0.53	0.32
E2	0.64	0.32	0.79	0.31	0.62	0.31
LogE1	0.51	0.42	0.64	0.49	0.49	0.44
LogE2	0.75	0.66	0.84	0.72	0.70	0.67
AVE	254.07	486.30	-50.16	66.5	254.63	494.81
R^2	0.74	0.35	0.80	0.31	0.74	0.35
Pbias	0.27	0.42	-0.05	0.06	0.27	0.42
KGE	0.67	0.36	0.77	0.38	0.67	0.35

Using the 1990 land cover, the modelled streamflow (for optimal parameters A) is represented well for the calibration period with a logE2 value of 0.75. During the validation period this falls to 0.66. An E2 value of 0.64 is satisfactory for the calibration period but drops to 0.32 during model validation. Pbias and KGE are reported as 0.27 and 0.67 respectively for the calibration period and 0.42 and 0.36 for the validation period. This suggests that the simulation tends to overestimate streamflow (for low flows) and performs better for the calibration period than the validation period.

The Scenario B simulation, using the relaxed model parameter constraints, shows a stronger model performance overall for both the calibration and validation periods where logE2 is 0.84 and 0.72 for the calibration and validation periods respectively. Pbias drops substantially compared to the Scenario A simulation, exhibiting the model’s tendency to underestimate streamflow during the calibration period and slightly overestimate streamflow during the validation period. However, the Pbias values of both scenarios are close to an optimal score of zero, indicating strong model performance. KGE for Scenario B is 0.77 for the calibration period, but again it drops substantially to 0.38 during the model validation period. Finally, use of the 2018 SANLC data set as input (Scenario C) has little influence on model performance. Pbias remains the same as in Scenario A with similar KGE, E2 and logE2 performances. Therefore, Scenarios A and C perform worse than the relaxed parameters of Scenario B indicating that parameter constraints impact model performance more than the landuse input does.

4.4.1.3 Parameter sensitivity

A sensitivity analysis of the various model parameters revealed that flow routing influenced simulated streamflow the most, followed by fast groundwater delay, direct surface flow delay and the scaling factor for MPS (Figure 4.6). Model inaccuracies seem to be due primarily to flow-partitioning parameters (flow routing, fast groundwater delay and direct surface flow delay) and available soil-water capacity, which are inherently highly heterogeneous spatially.

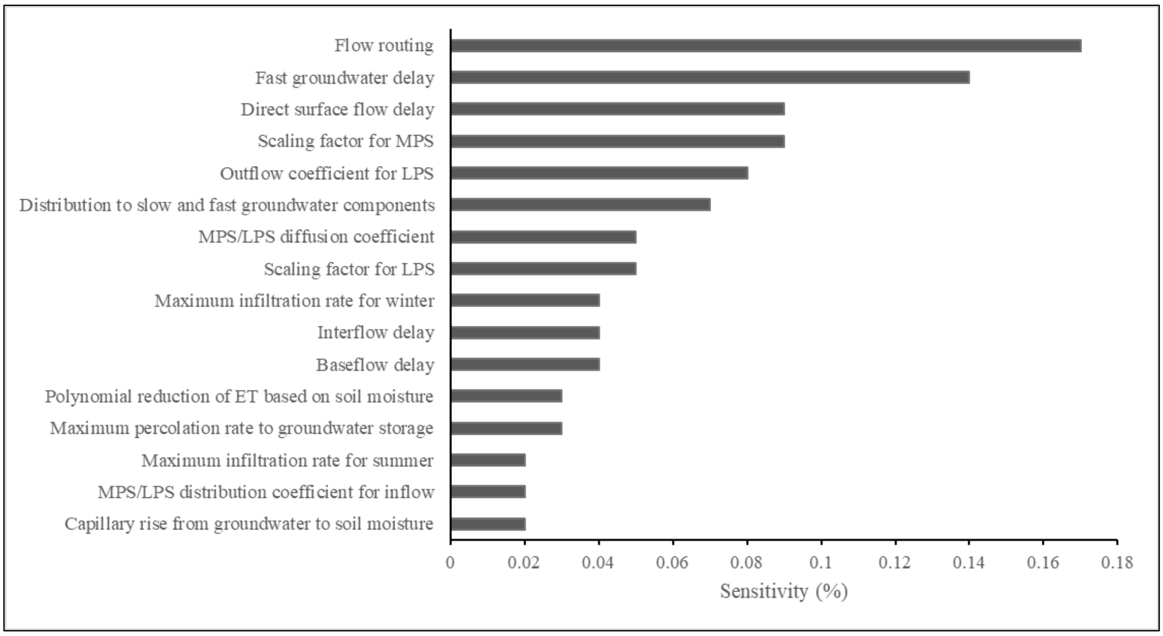


Figure 4.6 Regional sensitivity analysis of the Bot River model parameters.

A detailed representation of the most sensitive model parameter (flow routing) compared to the least sensitive parameter (capillary rise from groundwater to soil moisture) is presented in Figure 4.7.

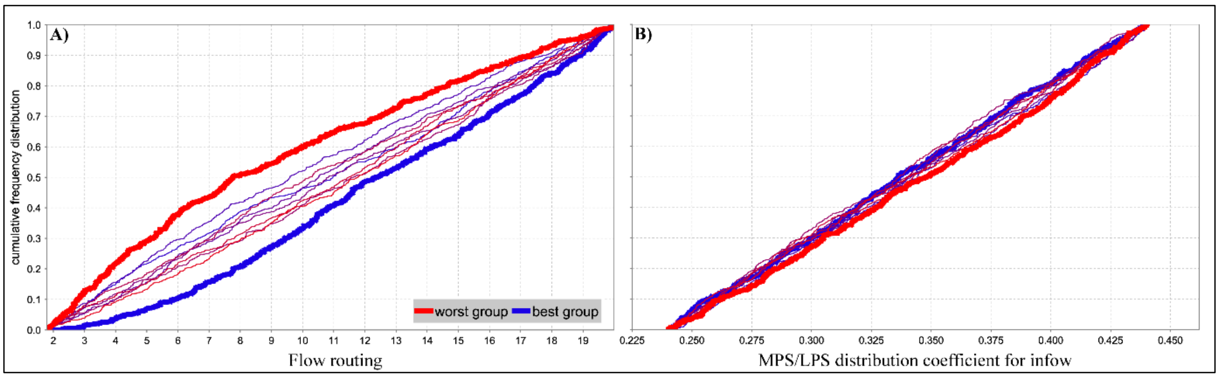


Figure 4.7 Regional sensitivity analysis for selected model parameters – (A) flow routing (most sensitive) and (B) the MPS/LPS distribution coefficient for inflow (least sensitive).

Parameters are regarded as insensitive when the difference between the best (blue line) and worst (red line) groups is minor. Thus, parameter sensitivity is principally determined by the deviations between the worst and best model groupings. For the flow routing, fast groundwater delay and direct surface flow delay parameters, there is a marked difference between the best and worst parameter groups, thus these parameters are highly sensitive and small changes to them affect simulated streamflows substantially. In comparison, parameters such as the MPS/LPS distribution coefficient for inflow show little difference between these groupings and have less impact on simulated streamflow.

4.4.2 Flow components

The mean monthly streamflow-generating flow components are displayed in Figure 4.8. There is a clear seasonal pattern to these flow components with direct surface runoff, interflow (soil water) and baseflow from both primary and secondary aquifers fluctuating during seasonal dry and wet cycles.

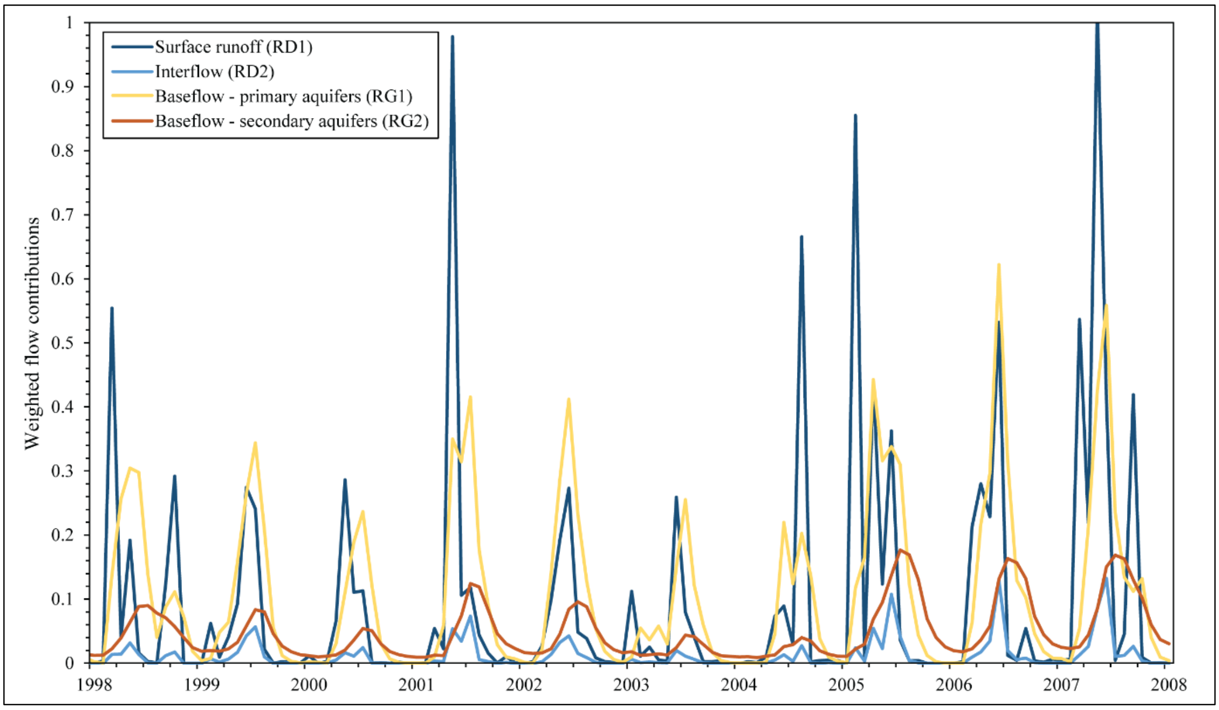


Figure 4.8 Monthly mean contributions of flow components to overall streamflow from model calibration to validation.

Direct surface runoff varies the most of all the flow parameters dropping to zero during dry months and increasing substantially during wet months. The second most significant flow component is baseflow contributions from primary aquifers, that is pore spaces in geological substrate or unconsolidated sands. Interflow contributes the least of these flow components possibly due to the high clay content of the soils, which limits infiltration into the soil. There is a notable reduction in baseflow from both the primary and the secondary aquifers during the 2003-2004 period (likely due to a drought event in the Western Cape during those years).

4.4.3 Sediment yield

Sediment yield was simulated for the entire Bot River catchment and for individual HRUs. These outputs are reported at daily time steps and then aggregated to provide an estimate of mean annual sediment yield. Sediment yield simulations were run for three different land cover scenarios: the 1990 land cover (Sediment Yield Scenario X), 2018 land cover (Sediment Yield Scenario Y) and undisturbed (pre-agriculture), natural conditions (Sediment Yield Scenario Z). The modelled sediment yields for the Bot River catchment are listed in Table 4.5.

Table 4.5 Simulated annual mean catchment sediment yields (1998-2008).

Scenario	Catchment mean annual sediment delivery (tonnes)	Annual sediment yield per km^2 (t/km^2/year)
1990 land cover (Scenario X)	366 073	423
2018 land cover (Scenario Y)	423 823	490
Natural conditions (Scenario Z)	19 306	22
1990 land cover – high surface runoff	388 194	449
1990 land cover – low surface runoff	366 619	424

Under natural conditions (Sediment Yield Scenario Z) the modelled sediment yield for the catchment is low at 22 t/km^2/year (Table 4.5). However, the agricultural impact on sediment delivery in the catchment is substantial. The 1990 land cover simulation (Sediment Yield Scenario X) predicts an annual sediment yield of 423 t/km^2/year, some 19 times greater than under natural conditions. The 2018 land cover (Sediment Yield Scenario Y) data resulted in the highest sediment output (490 t/km^2/year). Daily sediment yield for the Bot River catchment over the model calibration and validation periods is presented in Figure 4.9. Simulated sediment yield exhibits a similar pattern to catchment discharge and is greater in the winter months than during the dry summer. The greatest simulated sediment yield occurred on 6 October 2004 followed by 26 July 2007.

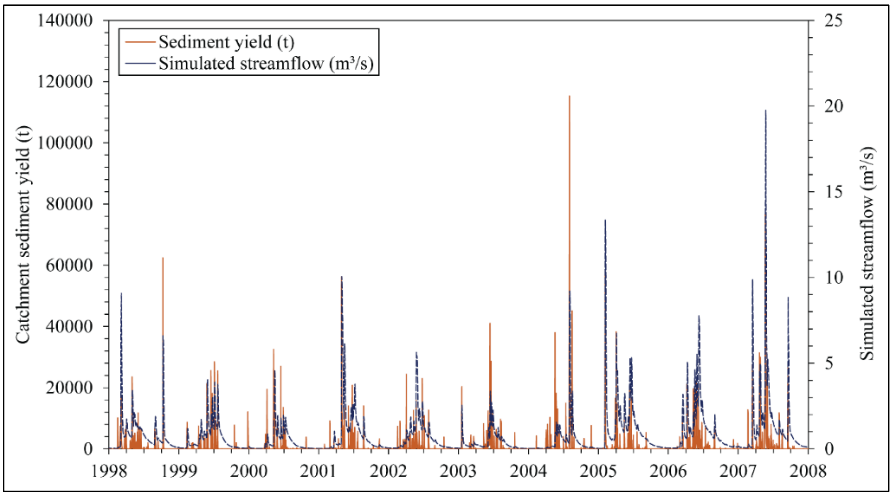


Figure 4.9 Daily sediment delivery in relation to simulated streamflow for the Bot River (Sediment Yield Scenario X).

Estimates of catchment soil erosion for the high and low direct surface runoff scenarios are at 449 and 424 t/km^2/year respectively (Table 4.6). The model parameters that differ the most between these two scenarios are fast groundwater delay and maximum infiltration rate for summer. This permits the model to simulate more direct surface runoff so leading to greater sediment yields – as anticipated for future climate change in the region.

Table 4.6 JAMS/J2000 model parameters and sediment responses for the high and low surface runoff scenarios.

Parameter	Scenario: High surface runoff	Scenario: Low surface runoff
Scaling factor for LPS	2.00	2.00
Scaling factor for MPS	1.99	2.00
Capillary rise from groundwater to soil moisture	0.89	0.31
Fast groundwater delay	10.00	0.14
Distribution to slow and fast groundwater components	1.00	1.00
Baseflow delay	1.83	1.24
Direct surface flow delay	2.20	2.20
Interflow delay	3.02	4.99
MPS/LPS diffusion coefficient	3.20	2.21
MPS/LPS distribution coefficient for inflow	6.62	4.39
Maximum infiltration rate for winter	122.30	120.91
Maximum infiltration rate for summer	169.02	227.33
Maximum percolation rate to groundwater storage	34.89	34.98
Outflow coefficient for LPS	9.99	9.98
Polynomial reduction of ET based on soil moisture	6.28	4.50
Flow routing	1.08	1.87
Model efficiency (E2)	0.51	0.69
Model efficiency (logE2)	0.71	0.76
Average daily fraction of direct runoff component (%)	0.24	0.13
Average daily fraction of interflow component (%)	0.12	0.02
Average daily fraction of lumped groundwater components (%)	0.64	0.85
Annual catchment sediment yield (t)	388 194	366 619
Annual catchment sediment yield (t/km^2)	449	424

These findings are corroborated by evaluating the impact of changing simulated direct surface runoff on sediment yield on an annual basis (Figure 4.10). When the annual mean daily direct surface runoff component increases, so does sediment yield for the catchment.

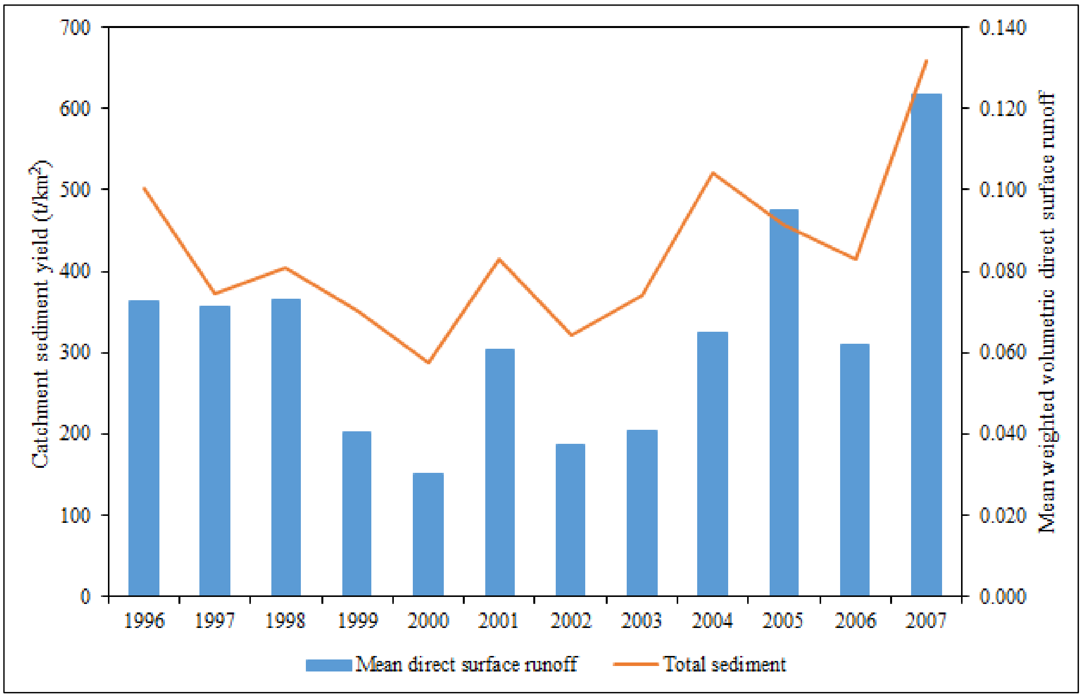


Figure 4.10 Impact of changing surface runoff on annual sediment yield in the Bot River catchment weighted by area.

The mean annual sediment yield, reported as tonnes per hectare (t/ha) per HRU (Figure 4.11), shows that sediment yield within the catchment is not spatially homogenous. It is evident that the mountainous regions contribute relatively little to erosion in comparison to lower-lying hillslopes. Conversely, soil erosion is more prevalent to the northern and central areas of the catchment that are dominated by agricultural land and soils with a higher erosivity.

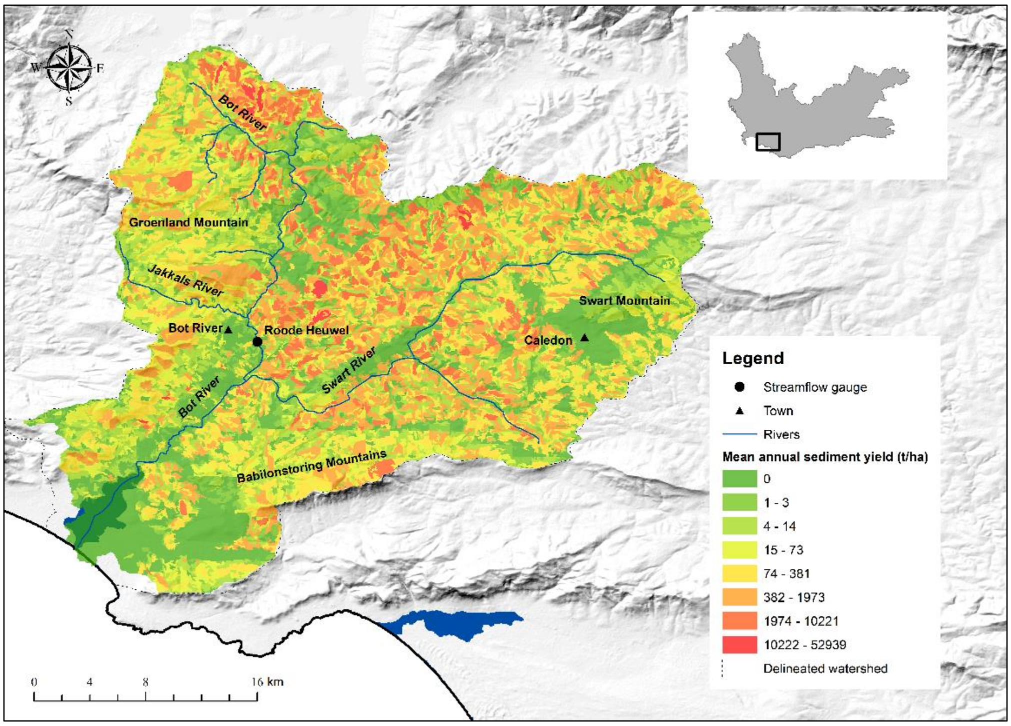


Figure 4.11 Mean annual sediment yield (t/ha) for each HRU for the Bot River, South Africa, based on a geometric interval classification.

Soil erosion in the catchment is driven by landuse (Figure 4.11 and Table 4.5) as well as the degree of direct surface runoff generated by the catchment (Figure 4.10 and Table 4.6). Agricultural transformation (Scenarios X and Y) results in substantially higher modelled sediment yield for the Bot River than natural conditions (Scenario Z), while a high direct surface runoff component scenario also predicts higher soil erosion than the low surface runoff scenario.

4.5 DISCUSSION

This chapter has explored the capabilities of JAMS/J2000 to model sediment yield under changing landuse conditions and climate variability, both inter-annual and over the long-term. Regarding overall stream dynamics, the C_v value of 0.54 for the Bot River represents moderate to high inter-annual flow variability, particularly for a perennial river. Flow variability is, however, below that of many southern African catchments which typically has a C_v value of 0.81 (Finlayson & McMahon 1988). The maximum discharge in April 2005 is most likely associated with a cut-off low pressure system that affected the southern Cape on 10-12 April (Holloway, Fortune & Chasi 2010). Indeed, many of the extreme streamflows in Western Cape rivers are driven by these cut-off low events (De Waal, Chapman & Kemp 2017). The model was limited by missing data in 2004 (Figure 4.12).

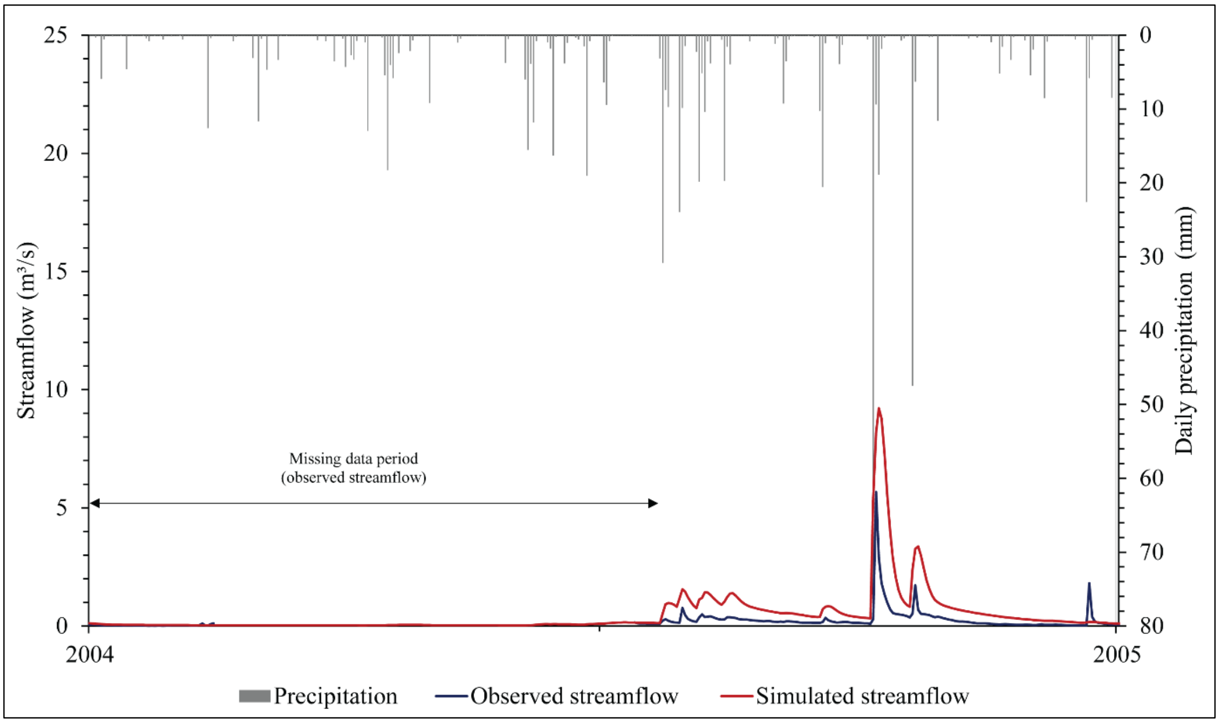


Figure 4.12 Missing observed data for 2004.

Although the winter months have higher flows than in summer, the maximum streamflow was observed in April, a boundary period. The transitions between seasons are times when instabilities often occur that lead to these severe events (De Waal, Chapman & Kemp 2017; Holloway, Fortune & Chasi 2010). The extreme event of 43.5 m^3/s on 27 July 2007 is quite likely associated with a strong frontal system characteristic of winter rainfalls in the Western Cape. It is noteworthy that observed streamflow data are missing for February to June 2004, so making an evaluation of the model for this period a difficult task (Figure 4.12). Furthermore, it is evident from visual inspection of the hydrograph (Figure 4.4) that streamflow during low-flow periods from 2003 to 2008 is artificially altered, for example observed streamflows suddenly increase or decrease with no observed meteorological inputs (discussed in Section 4.5.1).

The discussion below outlines the model performance and simulated flow components that contribute to the overall hydrograph (Section 4.5.1) as well as a detailed assessment of the modelled sediment erosion in comparison to other studies conducted in the region as well as the potential impacts of soil erosion on river systems in South Africa (Section 4.5.2).

4.5.1 Model performance and flow components

Despite the unavailability of some meteorological data for the catchment, the model performed well by most metrics (E1, E2, logE2, Pbias, KGE) for the calibration period (7 March 1998 to 31 December 2002). However, the model's efficiency drops substantially for the model validation period (1 January 2003 to 7 March 2008). Six factors may account for this. First, there are five months of missing observed streamflow data for the Roode Heuwel gauge during 2004 due to a gauge error. Second, the 2003-2008 period is characterised by several extreme rainfall events, such as 11 April 2005 and 27 July 2007, driven by cut-off low pressure systems and strong cold fronts. These events are underestimated by simulated flow, similar to the issues of modelling climatic extremes in the neighbouring Berg River catchment (Watson et al. 2022). Third, dramatic changes in observed streamflow during low-flow periods (summer) are not accounted for by the simulation. Explanations for this may be the abstraction of groundwater by local farmers, thus lowering the water table and subsequent baseflow contributions to the river, as well as direct pumping from the Bot River itself. Much of the Western Cape was affected by drought during 2003 and 2004 (Araujo, Abiodun & Crespo 2016). A response to lower rainfall is often to abstract more from rivers and groundwater sources. Thus, the 2003 drought may have resulted in increased abstraction from the river and groundwater, resulting in the streamflow reductions. It is also possible that this practice, started during the drought, may have been continued as normal operating procedure even after the drought had broken. Fourth, observed streamflow may be affected by water storage in farm dams. Farm reservoirs have no release schedule and only overflow once each dam's storage capacity is exceeded. Thus, summer rainfalls are not necessarily reflected in the observed flow (but would be simulated) as a large proportion of runoff is captured in storage and never released farther downriver. Sixth, the ability of rainfall–runoff models to simulate low-flow conditions is often limited due to a model's assumptions of storage stationarity in the ground and soil water components.

It is evident that a relaxation of model constraints improve the model's efficiencies (Table 4.4). This points to the potential problem of generalising model parameters within a broad geomorphic region as done by Watson et al. (2018; 2019; 2020; 2021; 2021). Further work is needed to provide better regional estimates of model constraints – including maximum infiltration rates in summer and winter as well as soil storage. However, Scenario C shows little difference on model performance compared to Scenario A and a worse performance than Scenario B. This suggests that landuse change has a very limited effect on the overall model performance and that land cover has either not changed substantially enough to influence simulated discharge or that other drivers, such as rainfall depth-duration, drive streamflows more than landuse.

An assessment of the flow components of the model – surface runoff (RD1) and interflow (RD2) coupled with baseflow from primary (RG1) and secondary (RG2) aquifers – reveals a high signal to noise ratio, indicating an extremely tight water budget. There is a strong seasonal pattern to streamflow in the catchment, driven by winter rains in the Western Cape, South Africa. Concerning flow components, direct surface runoff is the greatest contributor to streamflow in the winter months. However, during November to April streamflow is maintained by baseflow contributions from primary and secondary aquifers. The clay-enriched subsoil (B horizon) (Fey 2010) is a noteworthy limiting factor to water percolation through the soil that results in relatively larger overland flow components when compared to infiltration and subsurface flows.

4.5.2 Sediment delivery

Sediment yield in the Bot River catchment is substantial and the impact of agricultural transformation on erosion is evident. In general terms, natural, long-term geological erosion rates characteristically increase from lowland areas (with low gradients) ($<10^{-4}$ to 0.01 mm soil/a), to moderate gradient hillslopes (0.001 to 1 mm soil/a) and steep tectonically active relief (0.1 to >10 mm soil/a) (Montgomery 2007). Cultivated land within these different terrains creates erosion at similar rates to those of mountainous topography (Montgomery 2007).

Whereas the mountainous regions of the Bot River catchment are underlain by Table Mountain Group sandstone, the agricultural lowlands are found on Bokkeveld shale-derived soils (Curtis 2013; Fey 2010). Shales typically have a low tensile strength and are thus susceptible to weathering (Beckedahl et al. 1988) and the Bokkeveld shales weather to form nutrient-rich, clayey soils that are more prone to erosion. When observing the landscape, multiple gullies are evident on hillslopes underlain by Bokkeveld shales. The higher than normal soil erosion in the low-lying zones of the catchment is consequently attributed to the erodibility of these soils, the lack of natural vegetation, an annual fallow cycle and moderate slope gradient. The timeline of sediment delivery in relation to simulated streamflow (Figure 4.9) shows that sediment delivery is directly affected by rainfall (and streamflow). It is noteworthy that the maximum sediment yield for the catchment did not take place on the maximum streamflow date (11 April 2005) possibly due to the maximum infiltration rate parameter being set seasonally.

4.5.2.1 Comparison of results with other South African studies

It should be noted that because of a lack of sediment data (e.g. total suspended solids or turbidity) to validate the model, these results are largely conceptual in nature but they are comparable with the local-level results gleaned from a variety of sources listed in Table 4.7. These catchments and regional estimates of sediment yield have been selected owing to their geographical proximity to the study area, thus representing similar (although not the same) soil, climate and landuse characteristics. The Le Roux et al. (2008), Msadala et al. (2010) and Rooseboom (1975) sources provide regional estimates of sediment yield across the country. Those germane to the Bot River catchment are reported here. Estimates of sediment yield for several analogous catchments are based on measured reservoir storage and sedimentation data over time (Msadala et al. 2010; Rooseboom 1975; Rooseboom et al. 1992).

The simulated sediment yield for the Bot River is similar to those observed for the Wemmershoek reservoir (although Wemmershoek has a substantially smaller catchment (125 km^2) and higher rainfall). Estimated sediment yield for the Bot River is higher than estimates for the Stettynskloof, Moordkuil and the Klipberg Dams. However, the catchment area of the Bot River is far greater than the catchments of these dams. Despite the differences, the total modelled sediment delivery estimates appear to be within an acceptable range when compared to other data.

Table 4.7 Comparison of JAMS/J2000 MUSLE output for Bot River with estimates of sediment delivery in analogous catchments.

Period or date of analysis	Catchment or river	Reservoir	Catchment area (km^2)	Annual sediment yield per km^2 (t/km^2/year)	Source
			Bot River model		
1998-2008	Bot River (natural conditions)	NA	865	22	NA
1998-2008	Bot River (1990 land cover)	NA	865	423	NA
1998-2008	Bot River (2018 land cover)	NA	865	490	NA
1998-2008	Bot River (1990 land cover/high surface runoff	NA	865	449	NA
1998-2008	Bot River (1990) landcover/low surface runoff	NA	865	424	NA
			Comparative regional analyses		
1975	Overberg sediment yield estimate	NA	Regional estimate	150	(Rooseboom 1975)
1992	Western Cape region (Region 8)	NA	Regional estimate	16-450	(Msadala et al. 2010)
2008	Bot River and surrounds	NA	Regional estimate	500-2500	(Le Roux et al. 2008)
			Comparative catchments or rivers		
1957-1984	Wemmershoek (Franschhoek, Western Cape)	Wemmershoek Dam	125	310	(Rooseboom et al. 1992)
2010	Wemmershoek (Franschhoek, Western Cape)	Wemmershoek Dam	86	450	(Msadala et al. 2010)
1954-1984	Stettynskloof (Franschhoek, Western Cape)	Stettynskloof Dam	55	54	(Rooseboom et al. 1992)
2010	Stettynskloof (Franschhoek, Western Cape)	Stettynskloof Dam	60	49	(Msadala et al. 2010)
1950-1985	Hoeks River (Worcester, Western Cape)	Moordkuil Dam	176	9	(Rooseboom et al. 1992)
1964-1983	Konings River (McGregor, Western Cape)	Klipberg Dam	54	1	(Rooseboom et al. 1992)

An assessment of overall trends in sediment yield in South Africa suggests that sediment yield is increasing across the country. For instance, Msadala et al. (2010) report that 2010 estimates of sediment yields were higher than comparative data generated for the Rooseboom et al. (1992) sediment yield map. This trend of increasing sediment yield is observable for the Bot River. Msadala et al. (2010) ascribe some of the high values of sediment yield for Region 8 (which covers the Bot River catchment) in part to the influence of veld fires which increase soil erodibility by reducing vegetation cover and making soils hydrophobic. The influence of veld fires on sediment yield was not considered in this study but it is recommended that it be considered in future work.

4.5.2.2 Impacts of increased sediment delivery in South African catchments

Sediment delivery to streams and rivers affects the hydrology and physico-chemical status of rivers (De Jonge et al. 2004; Schoumans et al. 2014) as well as impacting reservoir capacity and weir gauges through sedimentation. Sedimentation of gauging structures is problematic as streamflow estimates from gauges are based on river height (stage) and additional sediment in the riverbed can artificially augment this to result in poorer data quality and an alteration of the river's stage-discharge relationship (Tomkins 2014). In South African conditions, the widespread use of Crump-weir structures (Wessels & Rooseboom 2009a; 2009b) limits the influence of upstream sedimentation by allowing sediment to pass freely over the gauging structure. Downstream sedimentation of river channels, however, can influence the potential for back-flooding, thus compromising gauged data. For sharp-crested (V-notch) weirs, also used in South Africa, sedimentation in the stilling pond is a problem as it affects the stage-discharge relationship. Aggradation of sediments beneath bridges is problematic as these structures are characteristically designed for specific return levels. Furthermore, sedimentation in dams is deemed a significant concern because storage capacity is compromised, so reducing water availability during dry seasons (Morris 2020). Therefore, changes to sediment loads over time pose major challenges for water management and monitoring.

Soil erosion in agricultural areas is also linked to phosphorus delivery to rivers as phosphorus strongly sorbs to mobile colloidal particles such as clay (De Jonge et al. 2004), yet there is little information on sediment yield of unmonitored and data-poor catchments in the country. An approach like the one adopted here can assist in gaining a better understanding of hydrological and sediment dynamics in little researched catchments. Further research should involve the collection of sediment data for the validation of sediment models similar to those data presented by Grenfell & Ellery (2009). Changing sediment delivery from hillslopes to receiving waterbodies such as rivers also leads to increased pollution potential, particularly nutrient loading. Further analysis of nutrient dynamics in under-monitored systems such as the Bot River is also warranted, while the establishment of natural vegetation buffers and land management can substantially reduce sediment delivery to rivers (Morris 2020).

4.5.2.3 Looking to the future: Is climate change influencing sediment yield?

One of the often spoken about impacts of climate change is changes in precipitation delivery (Allen & Ingram 2002; Trenberth 2011). In particular, the capacity of air to hold water vapour is positively correlated with increasing atmospheric temperatures (roughly +7% per 1°C) (Martinkova & Kysely 2020; Trenberth 2011). This relationship leads to increased availability of total precipitable water in the air and increased intensity of rainfall events (Martinkova & Kysely 2020). In South Africa the observed record suggests that mean annual temperatures have risen by at least 0.98°C since the 1960s (Ziervogel et al. 2014) with a concomitant increase in extreme rainfall events. Several studies have highlighted increasing frequency and intensity of rainfall extremes in South Africa due to climatic change (De Waal, Chapman & Kemp 2017; Du Plessis & Burger 2015; Johnson, Smithers & Schulze 2021; Kruger & Nxumalo 2017). The erosive potential of rainfall is directly related to the intensity and duration of these precipitation events (Burt et al. 2016). Thus, under conditions of higher rainfall intensity one expects increased soil erosion. As demonstrated by the increased soil erosion modelled in Scenario Y when compared with Scenario X (and indeed Scenario Z), soil erosion is controlled by landuse. However, changes in the erosivity of precipitation also have great importance as demonstrated by the high and the low direct surface runoff scenarios (Table 4.6). The JAMS/J2000 model shows the ability to model soil erosion under different climate scenarios. These scenarios reveal that an increase in direct surface runoff of 13 to 24% (average daily fraction of total streamflow) potentially results in an increase of 6% in sediment yield for the catchment (assuming an infinite sediment supply). With increasing intensity of extreme rainfalls in the country, the proportion of direct surface runoff generated by storm events is expected to increase. Therefore, one would expect increased soil erosion in South Africa’s catchments. Despite the results of these two scenarios being impaired by model uncertainty (the selection of two parameter sets), the results can also be imputed to climate change which would also affect these flow components. The modelling of streamflows and sediment yield during these extreme events remains a formidable challenge in South Africa because, as noted here, the predictive capacity for streamflows during extreme events is low and South Africa is especially prone to marked climate variability. Temporal cycles such as El Niño and La Niña phases impact on atmospheric dynamics in the region resulting in substantial changes to rainfall depth, duration and frequency. The JAMS/J2000 hydrological model harbours potential to accommodate such scenarios.

4.6 CONCLUSION

In this study a fully distributed hydrological model for the Bot River catchment was implemented that allowed for various flow components (direct surface runoff, interflow and baseflow) to be simulated. An additional component to the JAMS/J2000 model, MUSLE, was used to estimate sediment yield for the catchment under various landuse and flow component scenarios which were compared to sediment yield data of other local studies. This method facilitated the simulation of hydrological and sediment dynamics in a data-poor environment. The overall model performance was good for the calibration period but it decreased during model validation, the decrease conceivably being a function of a number of extreme events and abstractions from the river. The main finding is that the Bot River has high inter-annual discharge variability with deviation from the long-term median of annual mean discharge fluctuating between -76% (in 1969) and +170% (2013) and a coefficient of variation (C_v) of

0.54. Direct surface runoff contributes greatly to streamflow during winter, but summer low-flows are maintained primarily by baseflow from primary aquifers. Modelled mean annual sediment yield for the catchment is 423 t/km^2/yr for 1990, increasing to 490 t/km^2/yr for 2018 landuse. For low surface runoff conditions, mean catchment erosion was at 424 t/km^2/year and for a high surface runoff scenario this increased by 6% to 449 t/km^2/year. The JAMS/J2000 model can be used to estimate soil erosion under various landuse change scenarios as well as anticipated climatic futures. Distributed hydrological and sediment yield models can greatly assist in our understanding of local sediment dynamics and change over time, especially in data-poor areas such as the Bot River. Despite limited high-resolution geological and soil data, the use of remotely sensed data such as the HWSD and DEMs coupled with meteorological observations holds much promise. The work presented here represents a novel approach using a fully distributed JAMS/J2000 hydrological model with MUSLE output in a data-deficient area of South Africa. The application can be replicated elsewhere. For greater validation of results, the collection of daily sediment data in rivers can corroborate the model's output.

The modelling results support the evidence given by Msadala et al. (2010) and Msadala & Basson (2017) suggesting that soil erosion in South Africa is increasing. The modelling showed that soil erosion in the catchment is much more prevalent in low-lying foothills, characterised by agricultural landuses, limited soil conservation practices and readily erodible soils, as opposed to areas of higher elevation in the catchment. It is evident that soil erosion is increasing in the catchment due to landuse change. This is particularly concerning in the context of global landuse pressures and a changing climate. Increased sediment yield can potentially affect water quality in rivers in terms of turbidity and through the delivery of nutrients from agricultural activities to receiving waterbodies. In particular, phosphorus is often sorbed to clay particles and can be delivered to the Bot River during larger storms thereby affecting the nutrient status of the river. Furthermore, gauging structures, such as weirs, can be affected by sedimentation that further affects the gauges' stage-discharge relationships, rendering streamflow rating curves with greater uncertainty. Notably, while landuse remains the first-order control of soil erosion in catchments, changes in the erosivity of precipitation are also of great importance. The modelling of streamflows and sediment yield during these extreme events is a challenging application of the JAMS/J2000 model given that the predictive capacity for streamflows during extreme events is low – a fertile field for future research.

This chapter has highlighted and quantified the influence of landuse change on soil erosion in the Bot River catchment. Our attention now turns to the impacts of agricultural landuses on water chemistry in the catchment.

CHAPTER 5: THE IMPACT OF AGRICULTURAL TRANSFORMATION ON WATER QUALITY IN A CRITICALLY ENDANGERED VEGETATION LANDSCAPE – A CASE STUDY IN THE BOT RIVER, SOUTH AFRICA

5.1 INTRODUCTION

Healthy river systems provide a multitude of benefits to society and the ecosystems they serve, including the provision of water, habitats for aquatic species and nutrient cycling (Costanza et al. 1997; Nelson et al. 2009; Watson et al. 2019). One of the key threats to healthy, functioning rivers globally is pollution from agriculture (Kominami & Lovell 2012; Merrington et al. 2002; Wang et al. 2018). Numerous studies have indicated that agriculture is a leading cause of water degradation globally (Evans et al. 2019) through the release of agrochemicals (Arellano-Aguilar et al. 2017; Horak, Horn & Pieters 2021) and nutrients (Sharpley et al. 2015; Wang et al. 2018; Nie et al. 2018) from adjacent hillslopes into rivers (Evans et al. 2019). In agricultural regions water quality degradation typically arises from the mobilisation of nitrogen (N)- and phosphorus (P)-based fertilisers and manure (Owen et al. 2012). Agriculture can also impact water quality by exacerbating sediment input into rivers, while simultaneously reducing flows through water extraction for irrigating crops (Bonsch et al. 2015; Evans et al. 2019).

In South Africa freshwater pollution from agricultural sources is of great concern, the eutrophication of river systems being especially problematic (De Villiers & Thiart 2007; Harding 2015; Van Ginkel 2011). Analyses of dissolved inorganic nitrogen and phosphorus using long-term water-quality monitoring data, reveal that nutrient levels in 95% of the largest river catchments in South Africa exceeded the recommended water-quality guidelines for plant life, with nutrient fluxes being associated with agricultural landuse (De Villiers & Thiart 2007). These high nutrient levels are principally derived from surface runoff and the leaching of nitrogen and phosphorus from fertilised soils. Nitrogen is readily mobilised in soils owing to the solubility of NO_3^- in particular (Cameron, Di & Moir 2013; Van Kessel, Clough & Van Groenigen 2009). Phosphorus movement mainly occurs via surface runoff but leaching can also cause substantial losses from soils (McDowell, Worth & Carrick 2021; Stuart & Lapworth 2016). The leaching rates of nitrogen and phosphorus typically increase when nutrient application has been persistent over extended periods (Robertson, Schiff & Ptacek 1998).

Inorganic nitrogen levels in South African rivers have varied considerably over time, with concentrations changing by a factor of ten between 1985 and 2017 (Griffin 2017). Median inorganic nitrogen levels in rivers for the country were around 0.20 mg N/l during the late 1980s but had fallen to around 0.02 mg N/l in 2010 (Griffin 2017). Dissolved phosphate levels in South Africa's rivers increased from 0.016 mg P/l in 1985 to a high of 0.036 mg P/l in 2007 but have been in decline since 2008 (Griffin 2017). The decline has resulted in contemporary phosphorus levels comparable to those of the 1980s and this is attributed to a slight decrease in fertiliser application since 2008, improvements to wastewater treatment facilities across the country and a major manufacturer of washing detergent in South Africa having removed builder phosphorus from its products in 2010 (Griffin 2017). This decline in nitrogen and phosphorus is in contrast to findings in international studies (Bouwman et al. 2005), where many watercourses in the United Kingdom, for example, show evidence of increased nutrient and fine sediment loads (Owen et al. 2012).

Despite the overall decline in nitrogen and phosphorus levels in South African rivers, substantial seasonal differences still occur. Seasonal fluxes of nitrogen and phosphorus levels in rivers are controlled by rainfall and by the degree of agricultural transformation in the catchment. Where more than 20% of a catchment has been transformed to agricultural landuse, high nitrogen and phosphorus flux values exist (De Villiers & Thiart 2007), although these are subject to inter-annual and long-term variability. Any evaluation of the impacts of agricultural activities on water quality requires long-term data sets with high temporal resolutions (Burt et al. 2010). This type of monitoring is especially important for evaluating seasonal fluxes in nutrient concentrations for the better structuring of agricultural management plans (Dixon, Smyth & Chiswell 1999). However, in many countries with inadequate financial resourcing, there is often limited information available that detail water quality in river courses (Alam et al. 2007; Anvari et al. 2009).

Site specific studies needed to develop target water quality range (TWQR) values are scarce in South Africa (De Villiers & Thiart 2007). Data detailing how nutrient levels in surface waters change along a river's course are limited, particularly for smaller rivers, which are often unmonitored or hosts to only one monitoring site. The national chemical monitoring programme (NCMP) (Department of Water Affairs and Forestry 2004) does not have the capacity to monitor smaller catchments at a high spatio-temporal resolution and the observation sites are often linked to Department of Water and Sanitation (DWS) streamflow gauges where ease of access is greater (Hohls et al. 2002; Van Niekerk, Silberbauer & Hohls 2009). Consequently, many systems are unmonitored or have limited observation points. Where water quality observation sites do exist, they are generally linked to infrastructure such as gauging weirs as opposed to high-impact or at-risk reaches of a river. Accordingly, large rivers, where severe impacts of eutrophication are seen, are given greater monitoring and analytical attention (Hohls et al. 2002), but these high-order streams likely represent the end point in the catchment system. There is a genuine need to evaluate smaller streams and rivers that might be the sources of nutrient inputs into larger rivers. This requires sampling and analyses of river reaches throughout a catchment. Data with high temporal and spatial resolutions can assist the identification of the more impacted river reaches and an understanding of the impacts of agricultural activities on water quality, thus informing the positioning of representative and effective monitoring sites in a river system.

This chapter presents and discusses the results of a year-long monitoring of a typical river system in an agricultural setting, namely the Bot River, Western Cape, South Africa. The aim was to establish a baseline of nutrient levels in the river, describe the spatio-temporal distribution of water-soluble nutrients and identify the potential sources thereof. The Bot River catchment was chosen because it is a small, under-monitored river located in a region that has experienced substantial agricultural transformation from its natural conditions (Curtis, Stirton & Muasya 2013; Rebelo et al. 2006; Winter, Prozesky & Esler 2007). The catchment is home to critically endangered renosterveld vegetation and the nutrient status of the river might be indicative of the influence of agriculture on other watercourses in the landscape where natural vegetation has been replaced by agricultural landuses. The results highlight the importance of selecting appropriate and representative monitoring sites for these rivers when budgetary constraints limit the number of points that can be monitored sustainably. The findings should also be applicable to similar catchments in the Western Cape and beyond as they demonstrate the magnitude of seasonal nutrient fluxes in the system and the response to rainfall inputs in the catchment as well as demonstrating the need for detailed assessments of a catchment to identify appropriate water-quality monitoring sites.

5.2 STUDY SITE

A delineation using shuttle radar topography mission (SRTM) data for this study estimates catchment size at 865 km^2. Major tributaries to the Bot River are the Swart River, which flows in from the east and the Jakkals River which has its headwaters in the mountainous region to the west of the Bot River (Figure 5.1). Elevation ranges from sea level at the estuary of the river (the Bot River lagoon) to 150 m in the lower foothills of the catchment and about 1000 m in the Groenlandberg Mountain. The Swart Mountain to the east of the catchment has an elevation of ~ 1000 m at its peak. This study monitored 15 sites along the Bot River and its tributaries (Figure 5.1). The river channels in the tributary mountain streams (see Sites 3, 5 and 10) are 1 to 2 m wide and <0.5 m deep and are characterised by mixed rock and alluvial substrates.

Similarly, the upper reaches of the Bot River have relatively narrow river channels which start to deepen near Site 6 (under winter conditions) to >0.5 m with mixed rock and alluvial riverbeds. At Site 13 the channel is wider (3 m wide) and >1 m deep under winter conditions. Here the river substrate is mainly alluvial. The channel widens and deepens considerably in the lower reaches of the river as the topography flattens and discharge increases (Site 15). At Site 15 the river channel is not clearly defined, is >10 m wide with an unknown depth. There is a wide floodplain in this area, often inundated during winter months. Farm reservoirs are located upstream of Sites 1, 2 and 3. While the river at Site 1 flows year-round, streamflows for Sites 2 and 3 are impacted by the upstream dams' design, which only allows water to flow when these dams are full, during the winter months.

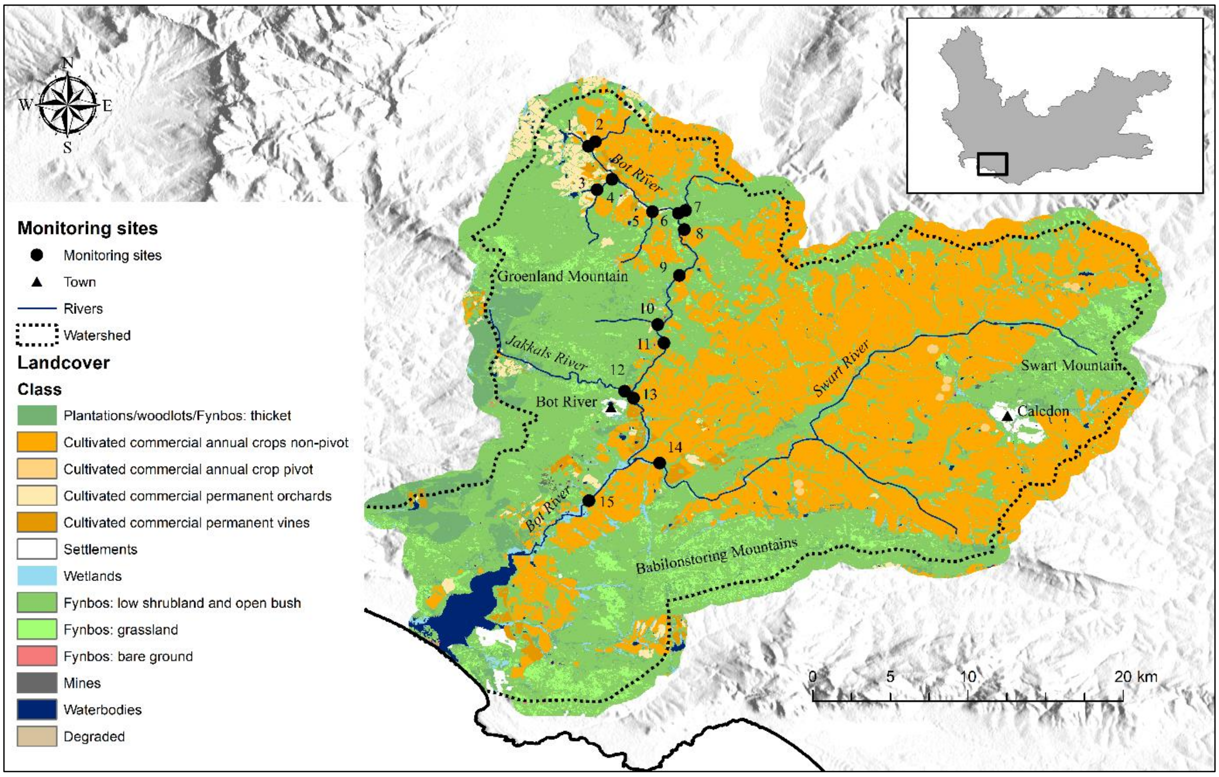


Figure 5.1 Bot River catchment with indicated monitoring sites.

Mean annual precipitation on the Groenland Mountain is 1100 mm/a and drops markedly to 450 mm/a at lower elevations toward the east of the catchment (Bailey & Pitman 2015). Most of the flow contribution to the Bot River stems from the Groenland Mountain tributaries in the west of the catchment. This area is located in a winter rainfall region that extends from May to October annually. The Köppen-Geiger climate classification for the catchment is Csa, with hot, dry summers (Beck et al. 2018). Mean monthly maximum temperatures range from 28°C in the hot summer months to 18°C in the winter, while mean monthly minima vary from 16°C in January and February to 7°C in July and August. June, July and August are the wettest months of the year, with ~70 mm precipitation in each of these months (Figure 5.2) in the centre of the catchment. Mean annual rainfall is ~ 450 mm. The average annual discharge at the DWS Roode Heuwel gauge (G4H014-A01) near the town Bot River is 22.6 x 10^6 m^3/a (Department of Water and Sanitation 2021), while mean catchment runoff (for the whole catchment) has previously been estimated at 116 x 10^6 m^3/a (Heydorn & Tinley 1980), although more recent estimates indicate a significant reduction of this figure to 65.9 x 10^6 m^3/a (Van Niekerk, Van der Merwe & Huizinga 2005).

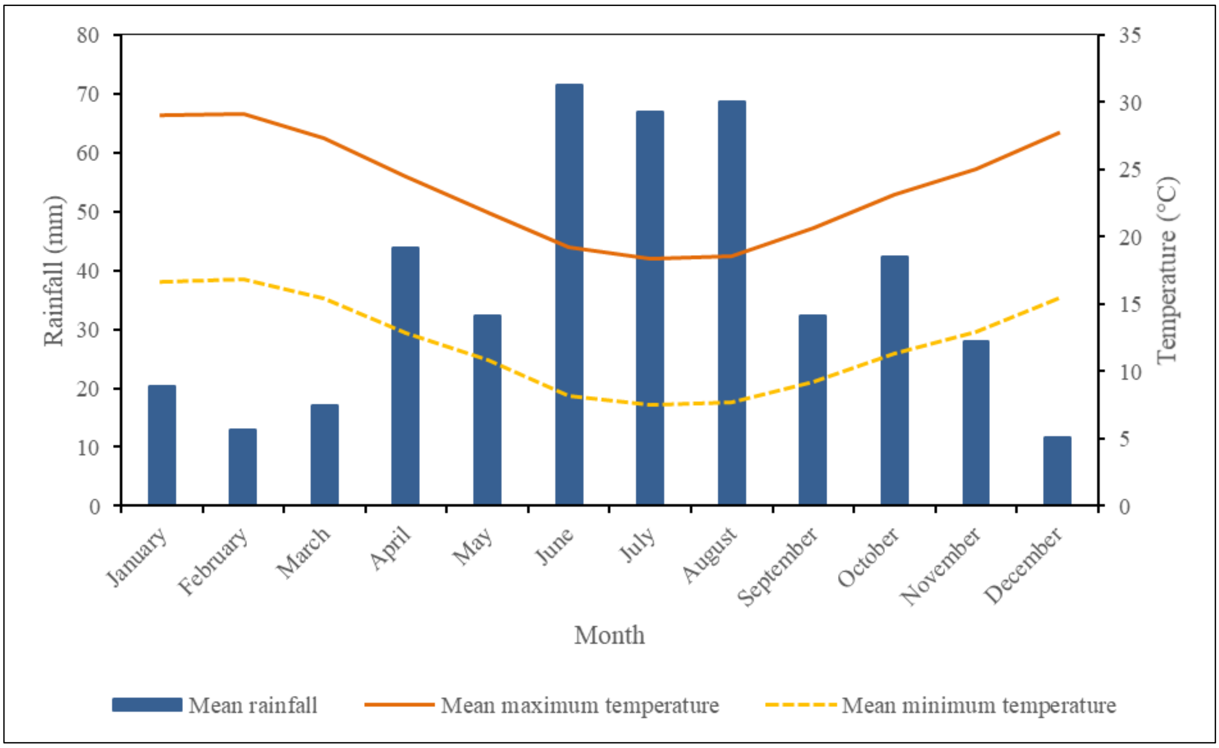


Figure 5.2 Mean monthly temperature and rainfall for Bot River (2002-2020) from the Agricultural Research Council's (ARC) meteorological station, Botrivier.

The underlying geology of the Bot River catchment consists mainly of Bokkeveld formation shales in low-lying areas and Table Mountain Group sandstone in the mountainous regions. The catchment mainly comprises lithic and duplex soils within the various landtypes (Fey 2010). Lithic soils in the Bot River catchment are characteristically shallow and stony and they erode to form convex slope crests and gentle footslopes (Fey 2010). The presence of duplex soils in the area indicates the enrichment of the subsoil with clay. Thus, there is a marked increase in clay composition in the B horizon when compared with the horizon above it (Fey 2010). The shale-derived clay soils in the catchment have been converted to agricultural use and they contribute relatively high inputs of silt and other erosion products from hillslopes to rivers in the area (Koop, Bally & McQuaid 1983).

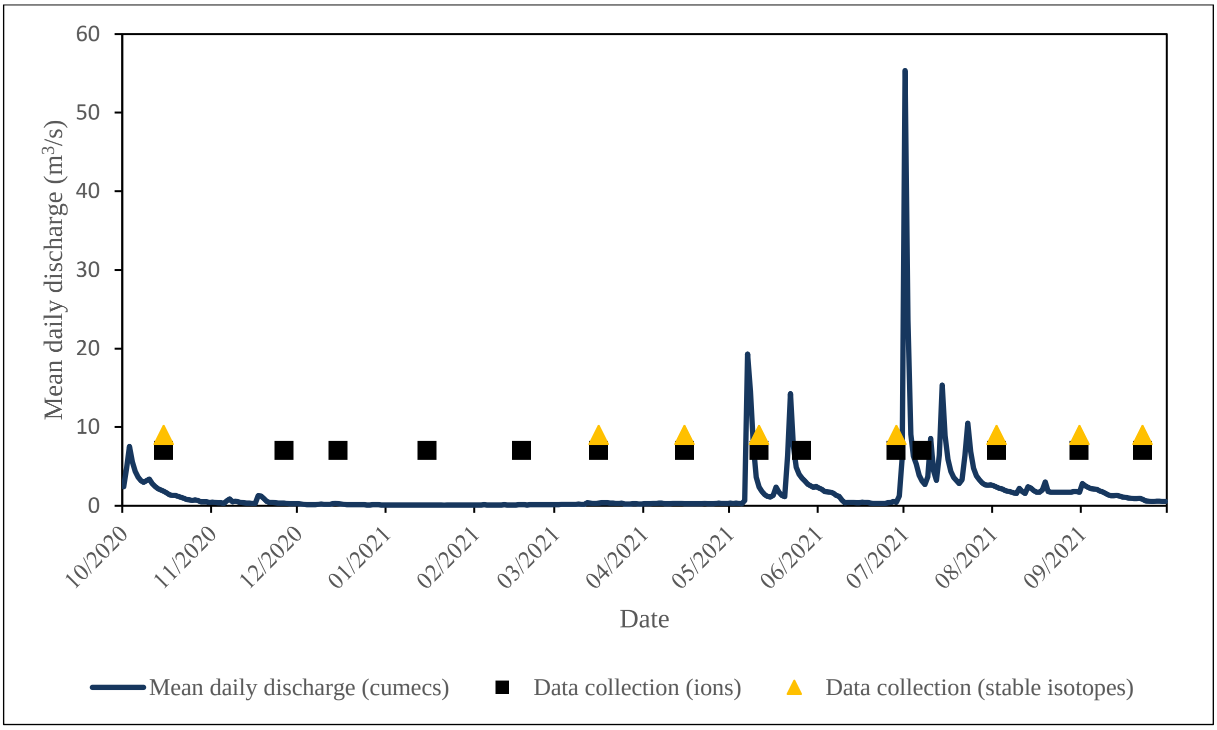
Shale renosterveld is the principal natural vegetation in the area, favouring nutrient-rich, shale-derived soils, while the nutrient-poor slopes of the mountainous areas are covered largely by mountain fynbos (Curtis & Bond 2013; Rebelo et al. 2006). As such, the shale tributaries of the Bot River contribute clear waters that contrast markedly from the waters of the Table Mountain Group-dominated mountain tributaries that are stained with humic acids. Landuse in the catchment is predominantly for agricultural activities. Extensive rainfed agriculture is present, dominated by oilseed and cereal crops that typically replace areas previously covered by renosterveld (Curtis 2016; Curtis & Bond 2013). The riparian zone of the Bot River is overgrown and heavily invaded by a mixture of invasive tree and plant species. Vegetation density near the river makes access to the channel extremely arduous in many places. Extensive reed beds of *Phragmites sp.* and *Scirpus sp.* can be found south of the Bot and Swart Rivers' confluence where the catchment gradient lowers substantially (Koop, Bally & McQuaid 1983) (Table 5.1).

5.3 METHODS

The overall approach in this study was to conduct a longitudinal assessment (year-long repeat sampling) of water chemistry (major anions and cations) in the Bot River. This allowed for baseline setting to future studies of nutrient content, the determination of the spatio-temporal variation in water chemistry and the delineation of river reaches that make major contributions to nutrient levels in the river. The selection of monitoring sites was determined by an examination of aerial imagery and the identification of major tributaries and access roads, and by conducting site visits to determine where access to the river was feasible. Sample analysis was done using several laboratory-based methods and the results were compared with target water quality ranges for aquatic systems in South Africa. This section provides a description of the sampling protocol used in the study (Section 5.3.1) and the analytical methods employed to assess water chemistry (Section 5.3.2).

5.3.1 Sampling protocol

On the basis of the above method, fifteen monitoring sites (Figure 5.1) were selected. Monitoring took place over a one-year period between 1 October 2020 and 30 September 2021. During drier months (November to April) one sample was taken at each site per month (n=6), while during the rainy season (May to October) samples were taken as soon as feasible after large rainfall events (n=8) (Figure 5.3). Samples were collected with a 50-ml syringe directly from the river reach at 10 cm depth and in the middle of the channel and then filtered through a 0.45-µm syringe filter into a clean 50-ml polypropylene tube. Each sample was immediately placed in a portable refrigerator and kept at approximately 2°C. Samples were transported to a laboratory where analysis took place within 24 hours of sample collection.



Source: Department of Water and Sanitation (2021)

Figure 5.3 Dates of data collection relative to streamflow at DWS gauge G4H014, Roode Heuwel.

To inform the sampling protocol, two river cross-sections between Sites 11 and 13 were analysed to measure whether there was any significant cross-sectional variation in water chemistry through the water column. Samples for these profiles were taken at various depth intervals (50 cm) and at 50-cm increments across the river's cross-profile. Because little difference was detected it was decided that a single sample per site, taken at the river thalweg at 10-cm depth, was sufficient as the river channel displayed a high degree of mixing. However, where water levels were >0.5 m depth at the river's thalweg, two samples were taken (surface and bottom) when it was safe to do so, that is if the river was flowing slowly enough to allow access to the thalweg without being swept away. This meant that samples were often collected two or three days post-storm peak. During the summer months some of the tributaries dried up, which is reported as no data (see Appendix B, Table B1). The specific characteristics of each site are given in Table 5.1.

It is noteworthy that the riparian zones of many reaches of the Bot River are heavily invaded by alien vegetation that makes access to the river challenging. River channel substrates are predominantly mixed, although alluvial channels are present in the lower course and where the slopes of foothill river reaches are low (Sites 1, 7 and 10).

Table 5.1 Specific site characteristics of each sampling point.

Site	Stream type	Channel substrate	Vegetation
1	Main stream – shale	Alluvial	In-stream reeds and grasses
2	Shale tributary	Rock	None
3	Mountain tributary	Mixed	Grasses and shrubs adjacent to river, no in-stream vegetation
4	Mountain tributary	Mixed	Alien vegetation (trees) in riparian zone, no in-stream vegetation
5	Mountain tributary	Mixed	Dense alien vegetation in riparian zone, no in-stream vegetation
6	Main stream	Mixed	Dense alien vegetation in riparian zone, no in-stream vegetation
7	Shale tributary	Alluvial	Mix of reeds and alien vegetation
8	Main stream	Mixed	Dense alien vegetation in riparian zone, no in-stream vegetation
9	Main stream	Mixed	Dense alien vegetation in riparian zone, no in-stream vegetation
10	Mountain tributary	Alluvial	In-stream grasses and reeds
11	Main stream	Mixed	Dense alien vegetation in riparian zone, no in-stream vegetation
12	Mountain tributary	Mixed	Alien vegetation in riparian zone, no in-stream vegetation
13	Main stream	Alluvial	Alien vegetation in riparian zone, no in-stream vegetation
14	Shale tributary	Alluvial	Extensive in-stream reeds
15	Main stream	Presumed alluvial	Extensive in-stream reeds

At most sites, in-stream vegetation is absent. However, in the lower course of the river, as topography flattens out, extensive reed beds are present in the river.

5.3.2 Analysis methods

A HANNA Instruments EC meter (model no. HI 5521, S/N: 04440013101) was used for laboratory electrical conductivity measurements. Anion and cation (Br^-, Cl^-, F^-, NO_2^-, NO_3^-, PO_4^{3-}, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+, K^+, Li^+, NH_4^+) concentrations were determined using ion-chromatography (IC) on a Metrohm 930 Compact IC Flex oven/SES/PP/DEG, in the BIOGRIP node for soil and water analysis at Stellenbosch University. An anion-exchange column (Metrosep A Supp 5 separation column of 4 x 100 mm) connected to an A Supp 5 guard column was used for ion separation. 20 µl of each sample were injected. The anion mobile phase containing 3 mM Na_2CO_3/2mM $NaHCO_3$ (1:1 ratio) was added at a flow rate of 0.8 ml/min. The column temperature for the anion analysis was set to 30°C. Detection of anions was performed using a conductivity detector. Cation analysis was performed using a Metrosep C6 separation column (4 x 250 mm) connected to a Metrosep C6 guard column, using 10 µl of each sample. The cation mobile phase containing 6.5 mM HNO_3 was introduced at a flow rate of 0.9 ml/min. The column temperature for the anion analysis was set to 55°C. Detection of cations was performed using a conductivity detector. Standard curves for all ions of interest were prepared using standards (certified reference materials) from De Bruyn Spectroscopic (South Africa) to quantify ions. All chemicals for preparing eluents were of the purest grade and were purchased from Merck. The lower detection limits and analytical precision for each ion's quality control (QC) sample is presented in Table 5.2. For calibration curves R^2 values for each ion's calibration was >0.999 and the % relative standard deviation (RSD) was <3%.

Table 5.2 Lower detection limit and quality control sample precision for each ion.

Ion	Lower detection limit (mg/l)	Quality control sample		
		Concentration (mg/l)	Mean % recovery	Standard deviation
Br^-	0.05	10	97	0.18
Cl^-	0.75	150	99	2.54
F^-	0.05	10	100	0.23
NO_2^-	0.05	10	95	0.32
NO_3^-	0.05	10	97	0.18
PO_4^{3-}	0.1	20	100	0.96
SO_4^{2-}	0.75	150	99	2.76
Ca^{2+}	0.75	150	98	3.33
K^+	0.05	10	98	0.39
Li^+	0.05	10	100	0.22
Mg^{2+}	0.375	75	99	1.57
Na^+	0.75	150	100	2.67
NH_4^+	0.05	10	99	0.5

Stable isotope analyses were performed at Stellenbosch University in the BIOGRIP node for soil and water. δ^2H and $\delta^{18}O$ ratios for each sample were measured using a Los Gatos Research (LGR) T-LWIA-45-EP, Canada model no. GLA431-TLWIA, stable isotope instrument. All samples were pre-filtered in the field using 0.45 µm filters and 1.9 ml of each sample were pipetted into 2 ml glass vials with polytetrafluoroethylene (PTFE) septum caps. Deionised water was used as a dummy sample to condition the system. QC was preserved by injecting a suite of water standards throughout each analysis (LGR working standards 1E, 3E and 5E). The three Los Gatos working standards, ten samples, and then another set of standards were run and repeated up to four times per batch. Water samples were injected into the heated septum port using a PAL LSI liquid auto-sampler. After injection, the heated water quickly vapourised and expanded into the laser cell of the liquid water isotope analyser. Each sample was injected nine times. The first four injections were disregarded due to memory effect, and the last five were averaged for reporting. The accuracy of internal QC standards was within 1.4‰ for δ^2H and 0.1‰ for $\delta^{18}O$. The precision of the GLA431-TLWIA analyser was ±1‰ for δ^2H and ±0.05‰ for $\delta^{18}O$. Data are reported in δ notation where:

$\delta = (R_{sample}/R_{standard} - 1) \times 1000$ Equation 3

and $R = {}^{18}O/{}^{16}O$ or ${}^{2}H/{}^{1}H$

Deuterium excess (d-excess) was calculated after Dansgaard (1964):

$d = \delta^{2}H - (8 \times \delta^{18}O)$ Equation 4

5.4 RESULTS

The results of the IC and stable isotope analyses are presented below. First, an analysis of river nutrient conditions is described followed by an overview of the stable isotopic composition results. A summary of the mean concentrations of all ions tested for each sample site is presented in Table 5.3, while the full data set for this study is available as Appendix B.

5.4.1 Water quality of the Bot River

The Bot River system is relatively brackish with high electrical conductivity (EC), Cl^- and Na^+ levels in particular. In terms of EC, the highest values occur at sample Sites 1, 7 and 14 (shale tributaries of the Bot River) while the mountain streams at Sites 4, 5, 10 and 12 show low EC (Figure 5.4). Site 3 is located downstream of a storage dam and shows a greater range of EC values. Sites located on the main trunk of the Bot River (6, 8, 9, 11, 13 and 15) show a much lower range in EC. EC between site 1 and 6 drops markedly in response to the input of mountain water from Sites 3, 4 and 5. The addition of water from Site 7 then slightly raises the EC of the river at Site 8. Furthermore, the substantial input from one of the major tributaries, the Jakkals River (Site 12), reduces the Bot River's EC between Sites 11 and 13. Finally, the substantial influence of the Swart River (Site 14) is again seen as EC rises at Site 15 in response to the extremely brackish water released from the Swart.

Where mountain streams were sampled (Sites 3, 4, 5, 10 and 12) the Cl^- and Na^+ concentrations are substantially lower than in streams that flow from a shale-dominated substrate (see Sites 1, 2, 7 and 14 for example) (Figure 5.4). The pattern of changing Cl^- concentrations is not temporally consistent across the different sites. Site 7, for instance, shows an increase in Cl^- concentrations only from May 2021 (the onset of the winter rains) whereas other sites (11 to 15) show more consistent Cl^- levels. Site 14, the Swart River, shows the highest Cl^- concentrations of all the monitored sites. Most of the Cl^- measurements fall within the range of concern described by Dugan et al. (2017) (230 mg/l) and the Canadian Water Quality Guidelines for the Protection of Aquatic Life (120 mg/l) for long-term (chronic) exposure (Canadian Council of Ministers of the Environment 2011) but exceedances at Sites 1, 7, 14 and 15 are common.

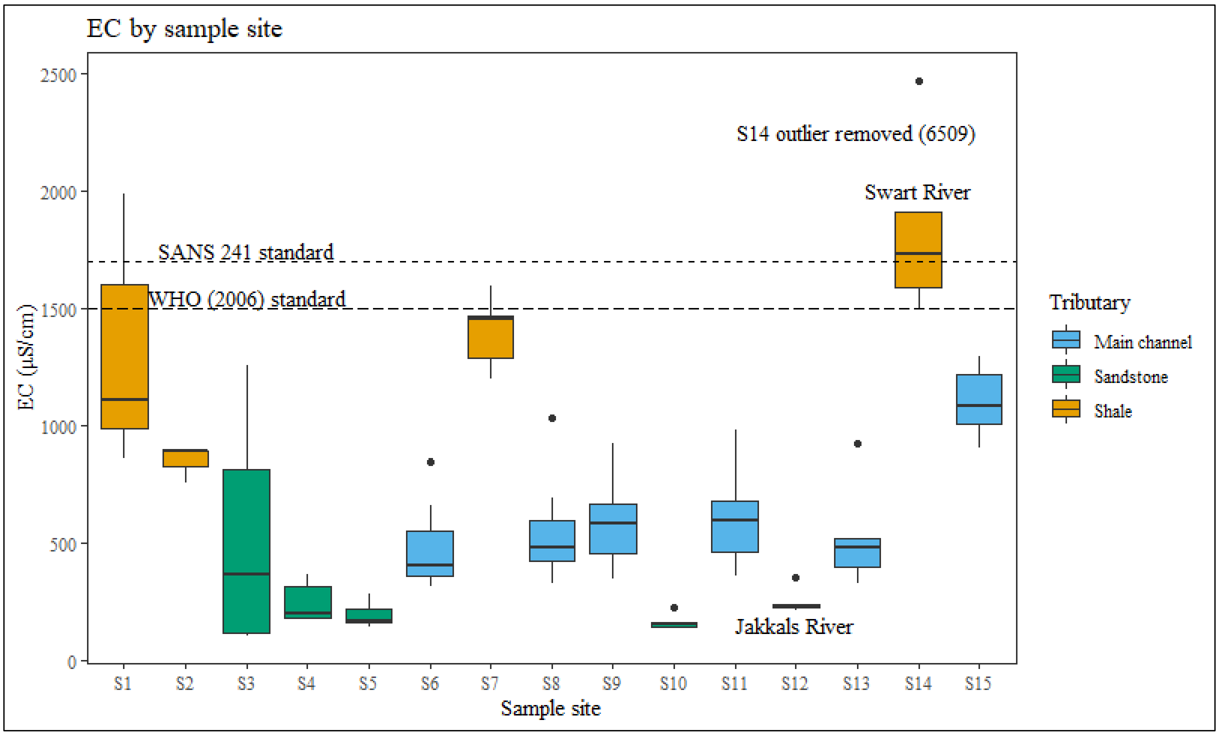


Figure 5.4 Analysis of electrical conductivity (EC) variation at each sample site along the Bot River with outlier for S14 removed to improve figure clarity and the World Health Organization (2006) and the South African National Standard 241 (South African Bureau of Standards 2015) guidelines for drinking water indicated.

Seasonal variations in both Cl^- and NO[x]-N concentrations are displayed in Figure 5.5. The results indicate periodic elevated levels of $NO_{[x]}$-N and Cl^- in the Bot River (indicated by exceedances over the dashed lines for $NO_{[x]}$-N and Cl^-) but these are site specific. Furthermore, there is a clear seasonal signature to nitrogen levels in particular, which increase by up to an order of magnitude during the winter months. While Cl^- concentrations show response over time, these are not synchronous with changes in $NO_{[x]}$-N.

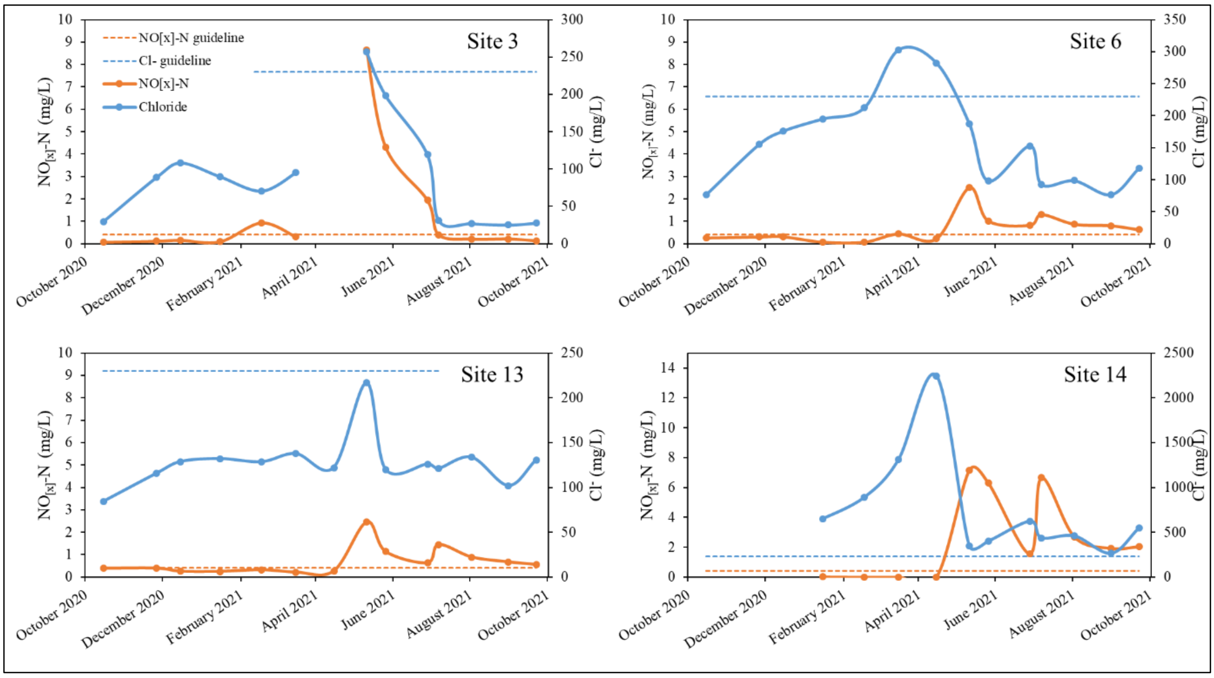


Figure 5.5 Chloride and $NO_{[x]}$-N fluctuations at selected sample sites along the Bot River with water quality guides for long-term exposure to chloride (Dugan et al. 2017) and NO[x}-N (De Villiers & Thiart 2007) indicated by dashed lines.

Sites 5 and 10 show no exceeding of the 400 µg N/l level, but the remaining sites all manifest elevated nitrogen levels during the winter months. All the sites (except Site 10) show a spike in nitrogen concentrations during May. From that point on, nitrogen levels remained elevated in the system until the dry season.

Dry conditions result in relatively little nitrogen being delivered into the river, while rainfall–runoff processes demonstrably result in elevated nitrogen in the system. These data are presented spatially in Figures 5.6A (dry months) and 5.6B (wet months). Again, the seasonal difference in nutrient loading of the river is clear.

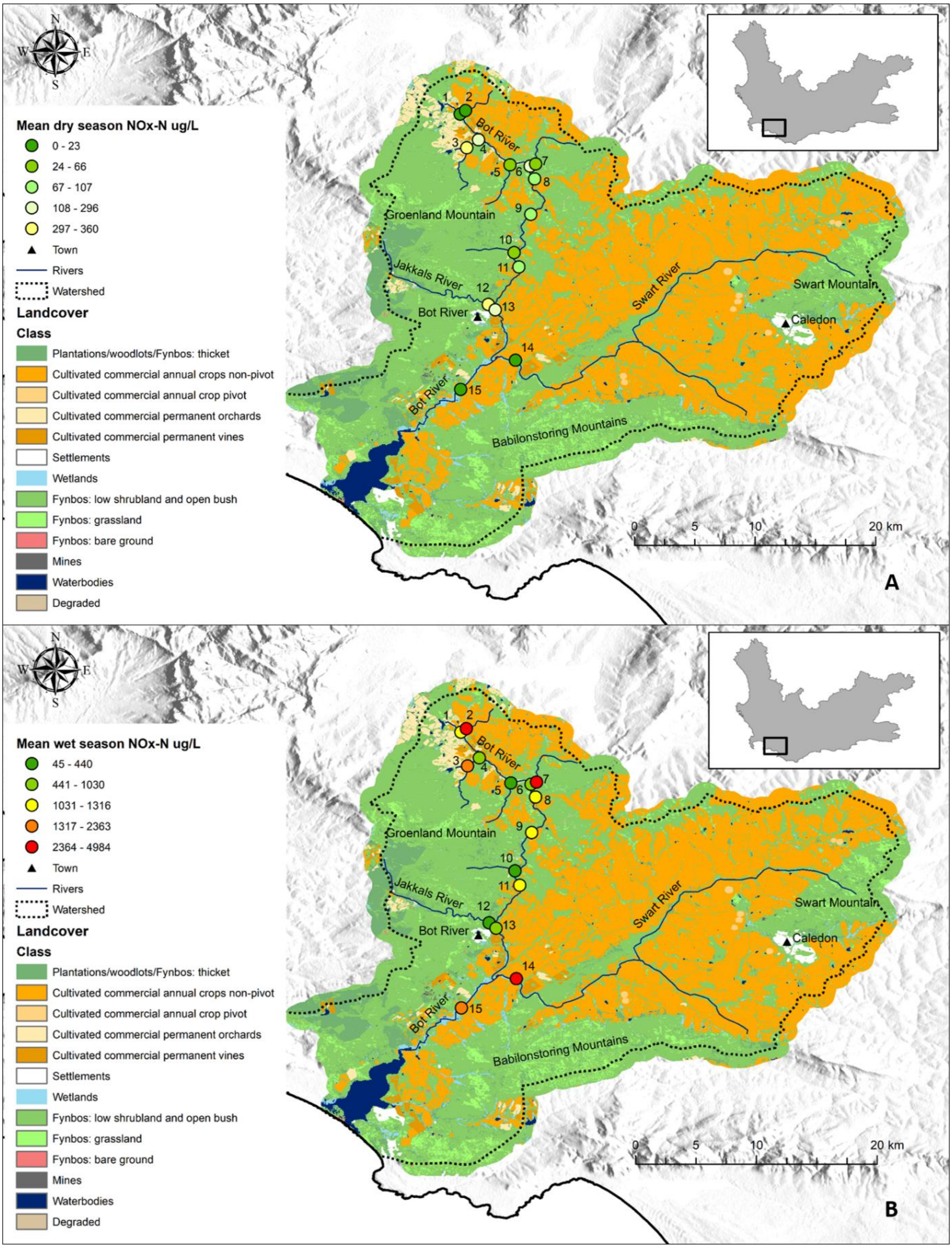


Figure 5.6 Seasonal mean $NO_{[x]}$-N (µg/l) along the Bot River for the dry season (A) and wet season (B).

The nitrogen concentrations in the Bot River not only vary seasonally but also over longer time frames. When the concentration of various ions at samples Sites 12 and 13 are compared with historic measurements (Bally & McQuaid 1985), it is evident that nitrogen concentrations in the river have increased markedly (an order of magnitude or greater) over the past four decades (Table 5.3).

Table 5.3 Summary of ion-chromatography (IC) results for anions and cations (annual mean concentration across all samples) at various Bot River sample sites in comparison to those taken by Bally & McQuaid (1985).

Ion (mean)	Lower detection limit (mg/l)	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11	Site 12 (Jakkals River)	Jakkals River (Bally & McQuaid, 1985)	Site 13 (Bot River)	Bot River (Bally & McQuaid, 1985)	Site 14	Site 15
F^- (mg/l)	0.05	0.17	0.20	0.09	0.06	0.06	0.11	0.19	0.10	0.11	0.03	0.09	0.01 ± 0.02	0.07 ± 0.07	0.07 ± 0.03	0.09 ± 0.08	0.92	0.19
Cl^- (mg/l)	0.75	336	212	89.8	67.4	81.1	159	270	154	173	54.1	163	63.47 ± 7.6	108.39 ± 58.54	129 ± 27.9	134.46 ± 37.9	742	280
SO_4^{2-} (mg/l)	0.75	40.9	52.2	22.9	22.6	12.1	31.3	35.8	27.9	28.3	5.92	27.9	8.19 ± 1.7	11.46 ± 6.92	22.3 ± 6.9	26.67 ± 10.86	57.0	31.1
PO_4^{3-} (mg/l)	0.1	ND*	0.06	ND	ND	ND	ND	ND	ND	ND	0.01	ND	ND	0.02 ± 0.03	ND	0.02 ± 0.03	0.43	0.04
PO_4^{3-}-P (mg/l)	n/a	ND	0.02	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.01	ND	0.01	0.14	0.01
PO_4^{3-}-P (µg/l)	n/a	ND	20.0	ND	ND	ND	ND	ND	ND	ND	4.26	ND	ND	6.52	ND	6.52	141	12.6
NO_2^- (mg/l)	0.05	ND	0.11	0.02	ND	ND	ND	0.01	ND	ND	ND	ND	ND	n/a**	ND	n/a	0.02	0.03
NO_2^--N (mg/l)	n/a	ND	0.03	0.01	ND	ND	ND	0	ND	ND	ND	ND	ND	n/a	ND	n/a	0.01	0.01
Br^- (mg/l)	0.05	1.14	0.55	0.31	0.19	0.27	0.49	0.85	0.47	0.52	0.17	0.49	0.20 ± 0.04	n/a	0.39 ± 0.08	n/a	2.57	0.96
NO_3^- (mg/l)	0.05	3.35	21.9	5.95	3.66	0.36	3.06	11.8	3.48	3.45	0.22	3.16	1.80 ± 0.76	0.03 ± 0.02	3.13 ± 2.65	0.27 ± 0.10	11.4	6.62
NO_3^--N (mg/l)	n/a	0.76	4.95	1.34	0.83	0.08	0.69	2.67	0.79	0.78	0.05	0.71	0.41 ± 0.17	0.007	0.71	0.06	2.57	1.50
$NO_{[x]}^-$-N (µg/l)	n/a	756	4984	1350	827	80.6	691	2679	785	778	49	714	406 ± 171	6.78	706	60.99	2579	1503
Li^+ (mg/l)	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.00 ± 0.01	n/a	ND	n/a	0.02	ND
Na^+ (mg/l)	0.75	161	108	46.6	35.7	41.1	81.1	145	79.1	87.9	27.5	83.4	31.4 ± 4.3	51.01 ± 26.05	65.8 ± 14.3	74.90 ± 21.69	418	155
NH_4^+ (mg/l)	0.05	ND	0.01	0.03	0.02	0.02	ND	ND	0.01	0.01	ND	0.01	ND	0.08 ± 0.04	0.00 ± 0.02	0.05 ± 0.03	0.14	0.02
K^+ (mg/l)	0.05	3.42	2.97	1.89	1.29	1.65	2.04	2.95	1.89	2.01	1.44	2.16	0.99 ± 0.23	0.85 ± 0.35	1.69 ± 0.5	2.45 ± 0.76	28.1	8.14
Mg^{2+} (mg/l)	0.38	39.7	25.9	9.62	7.62	7.84	16.5	29.1	16.6	17.7	3.62	15.9	5.54 ± 1.05	9.08 ± 5.08	12.4 ± 3.1	11.39 ± 3.03	62.6	26.2
Ca^{2+} (mg/l)	0.75	34.6	27.7	10.8	6.86	6.74	12.5	21.9	12.8	13.4	4.20	12.3	4.68 ± 1.09	4.33 ± 2.03	9.65 ± 3.2	7.20 ± 2.55	41.2	21.5

*Not detected (ND)

**Not available (n/a)

A comparison of the $NO_{[x]}$-N results for Site 13 and the long-term DWS monitoring of the same river reach (Department of Water and Sanitation n.d.) is presented in Figure 5.7. The data reveal high variability in nutrient loads (nitrogen) in the river that is linked to rainfall seasonality, while the mean for the study is greater than the long-term mean, although this difference in means is not statistically significant.

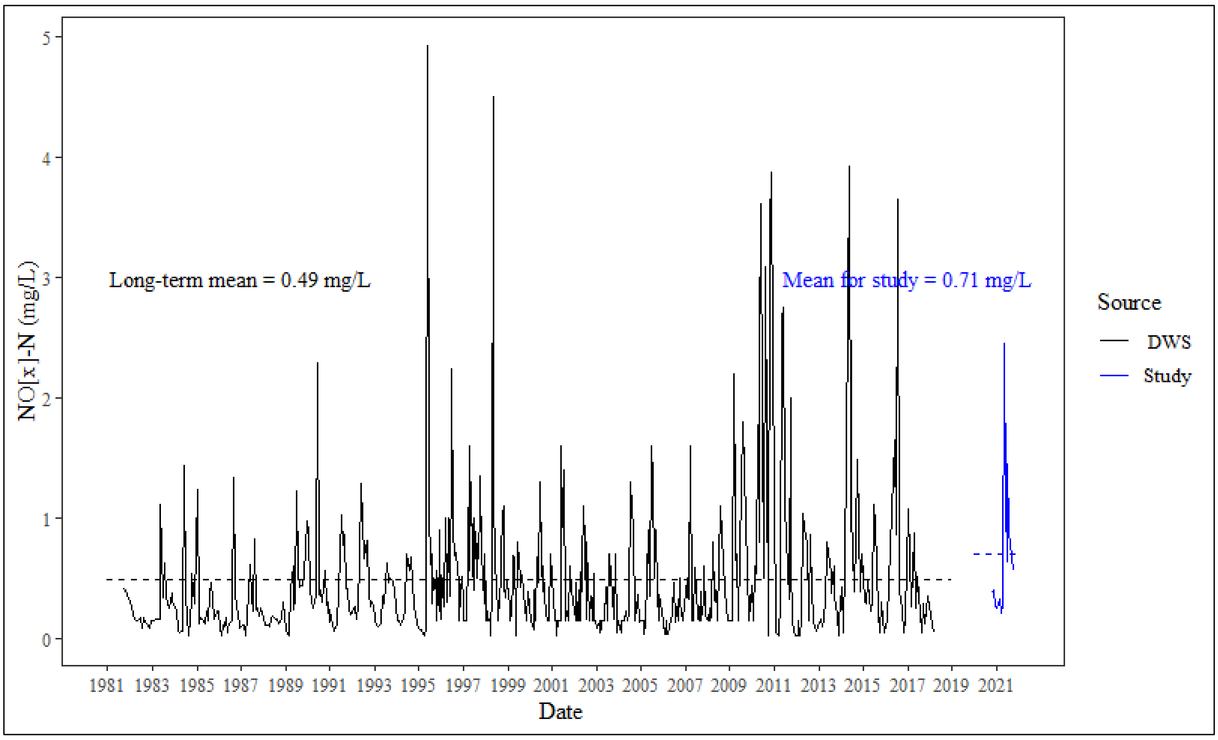


Figure 5.7 Long-term NO[x]-N data from site 13 in comparison with those collected during the study for the Bot River with data means depicted by dashed lines.

The Swart River showed high levels of soluble reactive phosphorus (SRP) for much of the wet season, but not in the dry season (Table 5.4). When SRP was detected, it exceeded the 100 µg P/l level suggested by De Villiers and Thiart (2007) for the protection of aquatic life.

Table 5.4 Soluble reactive phosphorus (SRP) for Site 14, Swart River.

Date	PO_4^{3-}-P (µg/l)	Date	PO_4^{3-}-P (µg/l)
15/10/2020	n/a	11/05/2021	263
26/11/2020	n/a	26/05/2021	172
15/12/2020	n/a	28/06/2021	478
15/01/2021	ND	07/07/2021	187
17/02/2021	138	02/08/2021	104
16/03/2021	ND	31/08/2021	ND
15/04/2021	ND	22/09/2021	210

*Not available (n/a)

**Not detectable (ND)

SRP concentrations are evidently higher from May to September when compared to the dry months of January to April and threshold exceedances represent a water-quality management concern in the Swart River.

5.4.2 Stable water isotopes

The bulk stable isotopic composition of river and rainfall samples are plotted in Figure 5.8. While the Bot River's local meteoric water line (LMWL) displays a very similar gradient to the global meteoric water line (GMWL) (Craig 1961), it is characterised by a larger deuterium excess (d-excess). By comparison, the LMWL for Cape Town (Harris et al. 2010) has a lower gradient than that of the Bot River LMWL. The river samples display a different isotopic composition to the local rainfall, consistent with evaporation from the river.

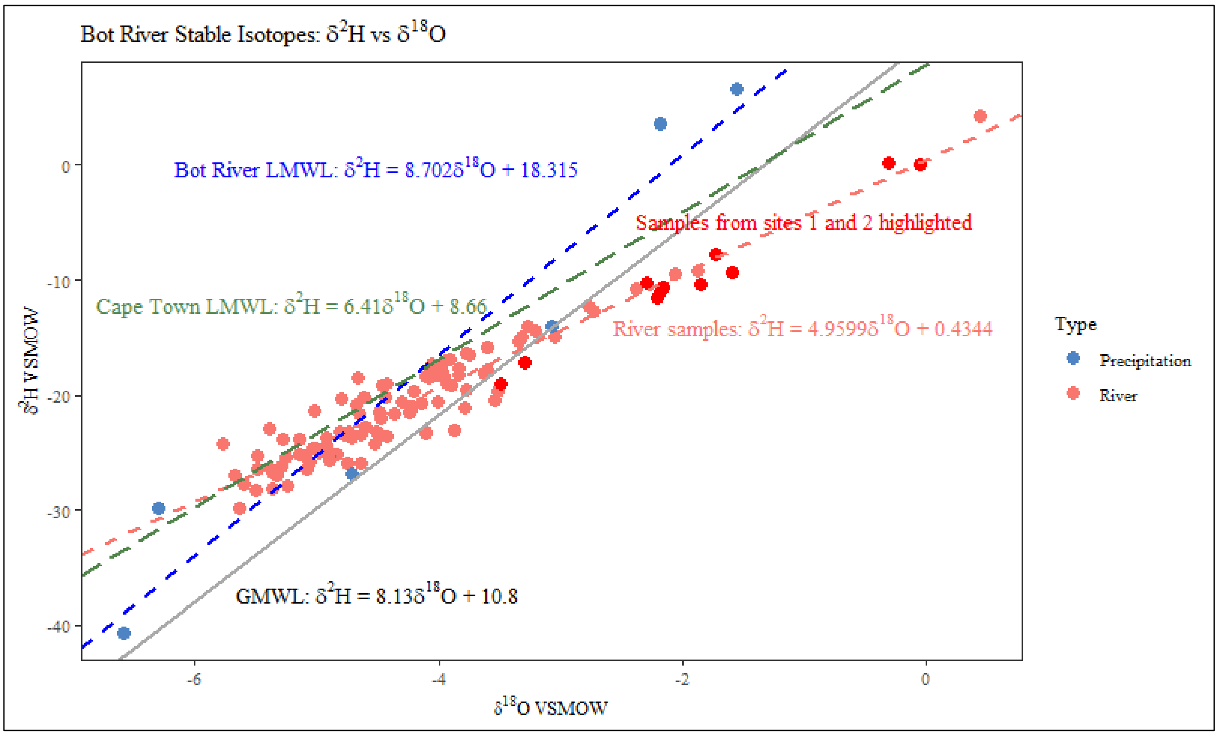


Figure 5.8 Bulk stable isotopic composition of Bot River samples and precipitation inputs from Site 7. Sites 1 and 2 are highlighted by red markers.

River samples demonstrate a distinct signal of evaporative loss which is consistent with the semi-arid conditions present in the Bot River catchment. D-excess from the LMWL by site is shown in Figure 5.9. D-excess is lower at Sites 1 and 2 while typically higher in mountain streams. Input from tributaries like Sites 4 and 5 dramatically increase the d-excess from Site 1 to Site 6, while the Swart River (Site 14) influences d-excess between Sites 13 and 15.

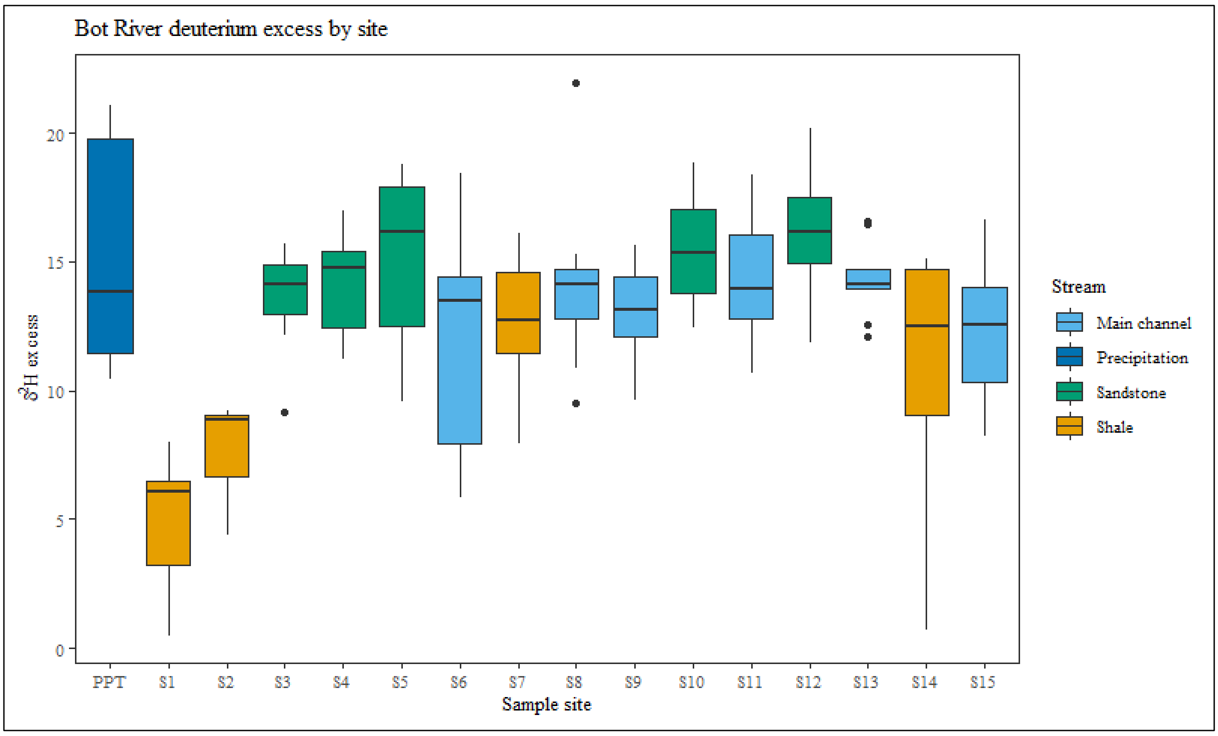


Figure 5.9 Deuterium excess for Bot River sample Sites 1-15 and precipitation samples.

Deuterium excess for Sites 3 to 5, 10 and 12 also show similar distributions to rainfall samples (PPT) and are markedly different from those of Sites 1, 2, 7 and 14 that stem from low-lying regions of the catchment.

5.5 DISCUSSION

The results presented above show some systematic variations in nutrient and stable isotope compositions across different sampling sites down the catchment, but these variations are not always consistent across different tracers. This extends to seasonality trends with some tracers showing a pronounced seasonality ($NO_{[x]}$-N), whereas for others the trend is less distinct (Cl^-). The interpretation of these results is therefore not straightforward and the intersection between different types of processes needs to be considered. This includes how anthropogenic processes, such as fertiliser application during agricultural cycles, intersect with natural processes such as evaporation during warmer summer months. In the following discussions the water quality challenges faced in the Bot River catchment are first considered (Section 5.5.1). Second, the relationship between landuse and nutrient loading in the Bot River is discussed (Section 5.5.2) followed by an evaluation of the meteorological inputs for the Bot River (Section 5.5.3). Finally, we reflect on the effective monitoring of water quality in a data-scarce catchment and the selection of appropriate monitoring sites for long-term observation and assessment (Section 5.5.4).

5.5.1 Water quality challenges in the Bot River catchment

This section discusses the spatio-temporal changes in salinity (Section 5.5.1.1) and the nutrient status (Section 5.5.1.2) of the Bot River over time.

5.5.1.1 River salinity

The Overberg region is characterised by high salinity of groundwater and surface waters relative to other areas in the Western Cape (Bugan, De Clercq & Jovanovic 2010; Mokoena et al. 2021). The Bot River system follows this trend with high EC and chloride concentrations in most of the sample sites. It is hypothesised that loose materials above underlying bedrock (Rûens shale) in the area comprise high levels of stored salts that have accrued via atmospheric deposition over time. Furthermore, vegetation types in lowland areas (renosterveld) differ from fynbos vegetation in the mountainous regions of the catchment. These variations in leaf structure can, for instance, influence the sequestration of wind-blown salts although this requires additional study. Thus a combination of geology, topography and vegetation characteristics results in higher EC and Cl^- values in lowland tributaries and inputs from foothill and lowland areas contribute greatly to overall salinity in the system. For instance, input of highly saline water at Site 7 increases the EC of the Bot River between Sites 6 and 8, whereas low-salinity water from the Jakkals River (Site 12), demonstrably lowers the EC and Cl^- concentrations from Site 11 to Site 13 (Figure 5.4). This supports the results of De Clercq, Fey & Jovanovic (2009) who found similar patterns in the Swartland region of the Western Cape – an analogous lowland renosterveld landscape. Salinity in the Bot River is site specific where it is based on underlying geology, which makes the selection of sample sites crucial when evaluating water-quality of a river.

Inconsistencies in the seasonal variations in chloride fluxes between sites can be caused by sites lower down the river course receiving inputs from a greater proportion of the catchment than those smaller subcatchments higher up in the drainage basin. For Sites 1, 7, 14 and 15 chronic chloride pollution is a potentially grave concern for the health of aquatic organisms. Salinity is also affected by site specific conditions. For instance, sites downstream of reservoirs (such as Site 1) show a different salinity pattern (high salinity in early winter) where summer evaporation from these storages and then spilling over in winter no doubt explains the marked increases in local salinity. The elevated chloride levels of Site 14, the Swart River, and their impact on the Bot River below its confluence (Site 15) is particularly concerning. However, the Overberg area is considered to have naturally brackish water and results should be considered in this context.

5.5.1.2 Nutrient loading

In general $NO_{[x]}$-N concentrations in summer are low and higher in winter (recall Figure 5.6). This is most likely tied to nitrogen being released from agricultural fertilisers applied in May (winter) in preparation for the growing season of wheat and canola crops. However, some subtleties in these trends, noted in the results, are worth discussing further. Although $NO_{[x]}$-N is higher in winter, the starting concentrations in summer are not consistent across all sites suggesting that microclimate and environmental context influence the $NO_{[x]}$-N systematics seen at different sampling sites. Little precipitation falls during summer and, as such, there is little hydrological connectivity between hillslopes and rivers and this limits nutrient inputs to the Bot River. Therefore, under dry conditions, nutrient inputs into the Bot River are low (Figures 5.5 and 5.6). In particular, $NO_{[x]}$-N concentrations are well below threshold levels of concern when considering the protection of aquatic organisms (De Villiers & Thiart 2007; Canadian Council of Ministers of the Environment 2012). Sites 3, 4 and 12 have the highest concentrations of $NO_{[x]}$-N during dry months. These sites are located on mountain streams that are fed by higher precipitation inputs from the higher elevation Groenland Mountain. During the dry summer months the mountainous regions above these sites experience some precipitation that is likely dissolving and transporting nutrients from hillslopes into the streams.

$NO_{[x]}$-N concentrations across the river system increase substantially during the wet winter months (Figures 5.5 and 5.6). Farmers in the region plant and apply NPK-based fertilisers during April and May in anticipation of the onset of the first rains. The samples collected on 11 May 2021 immediately followed a severe cut-off low pressure system that produced >120 mm rainfall over two days (according to measurements at a farm near Site 7) across the catchment. These were the first major rains of the winter season and represent first flush conditions for the catchment. Thus the availability of recently applied nutrients in the soils of adjacent hillslopes, coupled with significant rainfall, resulted in elevated nutrient levels in the river system from May onwards. These conditions persisted throughout the winter season until they began to decrease in October. The concentration-runoff profiles (Figure 5.5) represent diffuse nutrient sources, which generate seasonal concentration profiles that correlate with stream discharge (De Villiers & Thiart 2007).

A TWQR for inorganic nitrogen could not be established given the absence of information about local "unimpacted" conditions (concentrations) and insufficient historical nutrient data for each site. However, nutrient levels in the river were <0.3 mg N/l (oligotrophic) in 2004-2005 so classifying these waters as good quality (Herdien et al. 2006). However, in the present study all the sites except Sites 5 and 10 exceeded these levels, clearly indicating an increase in nitrogen levels since the early 2000s. Indeed $NO_{[x]}$-N concentrations were between 0.5 and 2.5 mg N/l for most sites (mesotrophic) for this study. While river conditions at Site 13 show elevated nitrogen during winter, the addition of water from the Swart River at Site 14 greatly degrades water quality further downstream in the Bot River. Inputs from mountain streams where there is less cultivated land show substantially lower levels of nitrogen.

SRP was below detectable limits for all sites except for Site 14, the Swart River. Orthophosphate in the Bot and Swart Rivers during 2004 and 2005 was reported as <0.05 mg P/l for the four sites analysed (Herdien et al. 2006). Thus, this study's results indicate little change, except for the Swart River which displays elevated SRP levels during the winter months (Table 5.4). When SRP was detected in the Swart River its level was above the recommended guidelines for aquatic ecosystems (De Villiers & Thiart 2007). For eutrophic conditions to develop, $NO_{[x]}$-N should be >400 µg/l coupled with PO_4-P concentrations of >20 µg/l (De Villiers & Thiart 2007). Phosphate is clearly a limiting variable (to biological growth) in the Bot River system. This, coupled with the fact that nitrogen levels are elevated in the cold winter months when algal growth is also slow, means that the river does not appear to be at great risk of eutrophication. Indeed, during the one-year monitoring of the system, no evidence of eutrophic conditions was seen. However, elevated nitrogen levels persist during the winter months and certain sites raise particular concern because they are linked to landuse having potentially negative impacts on local aquatic species (Dallas & Day 2004).

The data collected in this study provides a baseline for monitoring future changes in the Bot River system. The variations in the recommended guidelines given by many different sources for nutrient levels (Canadian Council of Ministers of the Environment 2011; 2012; Department of Water Affairs and Forestry 1996b; De Villiers & Thiart 2007; Dugan et al. 2017) brings into question whether these standards should be applied in an arid hydrology setting. Those presented by De Villiers & Thiart (2007) are likely the best reference for such conditions.

5.5.2 Is landuse driving nutrient levels?

There is substantial spatial variability in nitrogen levels along the river. Mountain streams consistently display lower nitrate levels than lowland areas. In particular, there is a strong link between landuse and nitrate concentrations in the river, with river reaches surrounded by cultivated, non-pivot agriculture (the Swart River) having substantially higher nitrogen levels than less transformed areas in the catchment. This again demonstrates the clear link between agriculture and nitrogen loading of river systems (Evans et al. 2019; Merrington et al. 2002; Wang et al. 2018). This is further made evident by the seasonality of nutrient fluxes in the system. Nitrogen levels increase by an order of magnitude or greater in response to winter rainfalls when compared to the dry summer months. This is directly related to the fertilisation of agricultural fields in the area during May and is seen in the spatial distribution of nutrient loads in the river. Here there is a clear association between surrounding agricultural landuse and nutrient concentrations when more transformed areas (such as the Swart River) are responsible for a greater degradation of water quality.

Regarding phosphorus, the Swart River is a major point source of phosphorus input into the Bot River. This too shows the impact of landuse on nutrient levels in the catchment. The Swart River drains an area dominated by non-pivot, rainfed agriculture (primarily wheat and oilseed crops) that has been highly transformed from natural vegetation. The high levels of phosphate and nitrate in this tributary are thus not very surprising.

5.5.3 Meteorological inputs for the Bot River

The greater deuterium excess present in the LMWL compared to the GMWL suggests that rainfall in the Bot River system is coming from an arid vapour source. This is typical of rainfall originating from the cold Southern Ocean, as is the case in the Western Cape. When considering the deuterium excess of all the sample sites from the LMWL for the wet and dry seasons it is noteworthy that not all of the water in the river is derived from local rainfall, indicating groundwater inputs into the system. Sites 1 and 2 show a marked difference from local precipitation inputs, indicating some form of secondary evaporation. These river reaches are downstream of farm storage dams with no input during the dry summer months and are being subjected to substantial evaporation.

Deuterium excess convergence further downstream is quite likely attributable to fewer inputs into the system. The large outlier for Site 14, taken during the dry season, is indicative of extreme evaporation. The Swart River was not flowing at the time of the year the sample was taken and had no connectivity from upstream or to the Bot River. The site was effectively a large pool of water at that time of the year with no input of water from upstream and it was experiencing high evaporation rates.

5.5.4 Lessons for effective monitoring of river systems in South Africa

The findings of this study illustrate that nutrient dynamics in a river are site (reach) specific and that they are influenced by surrounding landuse and tributary inputs, among other factors. Consequently, appropriate monitoring of river systems requires more than just one observation site within a catchment. With 277 tertiary and 1945 quaternary delineated catchments across the country many catchments are under monitored, or not monitored at all, particularly those in smaller catchments in dryland areas for reasons of a lack of financial and human resources. For instance, although the Bot River has a flow and water quality monitoring site at Roode Heuwel (DWS site G4H014), the nearest monitoring site in the Overberg wheatlands is >70 km further east in the Sout River at Kykoedy (G5H008), the only monitoring site on this 160-km-long extensively transformed river. The selection of these appropriate and representative monitoring sites as well as expanding the monitoring network for these systems is thus imperative. The selection of observation sites is a complex task that must consider the specific objectives of the monitoring programme (Dixon, Smyth & Chiswell 1999). For instance, sample locations can be located based on proximity to point sources or inputs of pollution to a stream or their proximity to key water sources or reservoirs (Ward 1973), all of which can introduce monitoring bias (Dixon, Smyth & Chiswell 1999; Sanders et al. 1983). For the monitoring of water quality, these authors advocate an analysis of river systems down the entire long profile to aid in identifying highly impacted river reaches as well as sources of nutrient inputs from tributaries. A detailed catchment assessment (such as that performed in this study) should be conducted to inform and direct the selection of monitoring sites. For instance, insight gained in this study points to the advisability of a monitoring site in the Bot River below the confluence of the Swart River tributary that is a large source of nutrient loading. Observation sites downstream of potential point sources (such as low quality tributaries) and river reaches with high nutrient loads should be prioritised over easily accessible ones. More recent technological advances in automated water samplers may assist in this regard (Bowes, Smith & Neal 2009).

In South Africa, several resource monitoring and protection measures have been implemented to address freshwater quality issues (Department of Water and Sanitation 2019). Efforts to monitor river health include the NCMP that has data extended back to the 1970s for some sites and the South African river health programme (RHP) established in 1994 and replaced by the river ecostatus monitoring programme (REMP) (Department of Water and Sanitation 2019). The programmes were developed in recognition of the need for more detailed information on the condition of South Africa’s river ecosystems. In 1999 water quality data were available at 2068 sample monitoring stations in the country (Huizenga 2011). However, by 2009 monitoring by the NCMP was reduced to approximately 40 primary sites and about 660 secondary sites focusing on major river catchments, about half of which were sampled at approximately two-week intervals (Griffin, Palmer & Scherman 2014). This illustrates the deterioration of water quality monitoring over time due to budgetary constraints. The results presented here demonstrate that smaller systems are still subject to nutrient loading and degradation of water quality, thus requiring dedicated and sustained monitoring. The United Nations sustainable development goal (SDG) 6.3 for improving water quality by reducing pollution (United Nations 2022) also requires monitoring and reporting through the global environment monitoring system for freshwater (GEMS/Water) programme (United Nations Environment Programme 2022). As such, South Africa reports on SDG indicator 6.3.2 – the proportion of bodies of water with good ambient water quality – that 41-60% of the country’s waterbodies have good ambient water quality (United Nations Water 2022). However, under-monitoring in South African rivers limits the quality and completeness of these data provided and impacts our understanding of the global status of water quality. Unfortunately, budgetary constraints remain a significant limitation to the location, number and frequency of samples that can be taken and analysed. The use of modern sampling instruments such as portable, in situ instruments can reduce laboratory costs, while automated water samplers could be used in periods or high flow (and greater nutrient delivery to rivers). Site selection for monitoring demands a full analysis of the river to determine any problematic river reaches and prioritise these sites.

5.6 CONCLUSIONS

This research aimed to establish a baseline of nutrient levels in the Bot River by describing the spatio-temporal distribution of nutrients and determining the potential sources thereof. The NCMP has one monitoring location for the Bot River (at the Roode Heuwel gauge) while some other sites with ad hoc measurements exist. This study introduced higher spatial resolution data for the entire longitudinal profile of the river and determined that water quality is reach specific and that the Roode Heuwel site is not representative of the entire catchment and also underestimates the water quality issues at certain sites. Regarding water quality, the Bot River system is relatively brackish in nature, with high salinity and chloride levels present at many of the sampled sites. Salinity increases during winter months when rainfall dissolves and mobilises salts in the landscape but this is not synchronous with changes in $NO_{[x]}$-N. Elevated dissolved inorganic nitrogen is seasonally present in the Bot River system and persists from May to September each year. These levels are cause for deep concern as they approach or already exceed the thresholds for aquatic ecosystem health. The Bot River’s trophic level appears to have increased when compared to 1985 (Bally & McQuaid 1985) and 2004/2005 data (Herdien et al. 2006). SRP is limited in the Bot River with results all below detectable limits. However, the Swart River has episodic exceedances of the 100 µg P/l maximum recommended water quality guideline for aquatic life.

There is a clear link between landuse and nutrient levels in the catchment. Much of the land in the eastern region of the catchment has been transformed to wheat and canola fields and the application of fertilisers is consistent with the high nutrient load of the Swart River tributary. Mountain streams exhibit low nutrient levels and help to dilute nutrient levels in the Bot River downstream of their respective confluences with the main trunk of the Bot. There is clearly a need to remediate and limit nutrient inputs to rivers in the Overberg region from agricultural hillslopes.

The findings of this research demonstrate that the selection of appropriate and representative monitoring sites for rivers is crucial. Many rivers are under- or unmonitored and where observations sites do exist they are proximal to gauging infrastructure. Examination of river systems down the entire long profile before selecting sample sites for long-term monitoring will aid in identifying highly impacted river reaches as well as the sources of nutrient inputs from tributaries, which can then be targeted for observation. A detailed catchment assessment (such as that performed here) should be conducted to inform the selection of optimal monitoring sites. As budgetary constraints still affect the number of monitoring sites along South African rivers, the use of portable, in situ measuring instruments can reduce laboratory costs and certain periods of the year can be targeted – post-rainfall events for agricultural regions. Future investment in these types of technological advances would be welcome. Specific ions of importance for this study were Cl^-, NO_3^-, NO_2^- and NH_4^+. SRP was often below detectable limits and it is proposed that future analysis should consider total P in the water samples.

CHAPTER 6: EXPLORING THE POTENTIAL OF CRITICALLY ENDANGERED NATURAL VEGETATION BUFFERS TO MITIGATE FRESHWATER POLLUTION – A CASE STUDY IN THE RENOSTERVELD, SOUTH AFRICA

6.1 INTRODUCTION

Non-point surface runoff from agricultural hillslopes is considered the primary cause of nutrient loading in rivers and it can result in the degradation of water quality and eutrophication (Kominami & Lovell 2012; Wang et al. 2018; Zhang et al. 2010). Soil leaching and soluble nutrients in overland flow result in the delivery of nutrients to freshwater bodies (Chapman et al. 2005; Heathwaite, Quinn & Hewett 2005; Ulén & Mattsson 2003). Therefore, nearby river systems in agricultural landscapes often contain higher levels of nutrients (particularly nitrogen (N) and phosphorous (P)) than those under natural conditions (Wang et al. 2018). As such, cultivated hillslopes are a significant management concern when considering freshwater pollution (Barcelo 1997; Lam, Schmalz & Fohrer 2010; Smith & Siciliano 2015). Mitigation of nutrient delivery to rivers involves agricultural stewardship and pollution control (Kay, Edwards & Foulger 2009). Pollution from hillslopes can be mitigated through effective crop management practices and vegetation remediation (Kanwar, Colvin & Karlen 1997). One particular mitigation intervention is the use of vegetative buffers located between agricultural fields and riverine systems. These buffers demonstrably have the potential to reduce agricultural runoff and hence also nutrient loading of rivers and streams (Connolly et al. 2015; Petersen, Jovanovic & Grenfell 2020; Vought et al. 1994). Such buffers offer a valuable ecosystem service by filtering surface runoff and near-surface flow through the cycling of chemicals and nutrients within the root zone of plants (Schoumans et al. 2014).

Vegetation buffers can effectively remove suspended solids, nutrients and pesticides from surface runoff by slowing down overland flow, acting as deposition zones for eroded sediments and allowing runoff to infiltrate into the soils for remediation by plants. The effectiveness of buffer strips is determined by the characteristics of these buffers in terms of width, slope vegetation present, soil type and flow velocity among others (Liu, Zhang & Zhang 2008). Much research has been done on topics like the efficacy of vegetative buffer strips to remediate water contamination, appropriate buffer width and vegetation composition (Allaire et al. 2015; Borin et al. 2005; Kavian et al. 2018; Zhang et al. 2010). In many areas however, little is known about the ability of indigenous vegetation to remediate polluted waters (Petersen, Jovanovic & Grenfell 2020). Nonetheless, the determination of the ecosystem services potentially provided by indigenous vegetation, can greatly assist in the conservation of these habitats (O'Farrell, Donaldson & Hoffman 2009).

The Bot River catchment in South Africa is dominated by agricultural landuse where, typically, a wheatland–fallow (or other crops such as canola, cereals, lupines and lucerne) system is practiced (Herdien et al. 2006). The area is home to renosterveld vegetation, a highly endangered vegetation type that forms part of the Fynbos biome. Renosterveld vegetation is described as a grassy shrubland (Curtis & Bond 2013; Winter, Prozesky & Esler 2007). Due to landuse transformation from natural to cultivated fields over the past two centuries, only 4-10% of the original renosterveld's spatial extent remains, most of which is on privately owned land (Curtis & Bond 2013; Rebelo et al. 2006). Renosterveld vegetation is highly fragmented making the determination of effective and appropriate conservation and management measures a challenging task. This fragmentation has resulted in the spatial distribution of renosterveld to be largely limited to steep slopes, rocky soils and adjacent to rivers and stream lines where no ploughing has been possible (Figure 6.1).

Figure 6.1 A renosterveld fragment located on a steep, rocky outcrop surrounded by cultivated fields, Bot River, South Africa.

Although the results of a controlled laboratory analysis of the phytoremediation potential of renosterveld plant species demonstrate their high potential for removing nutrients (N and P) and pesticides (Jacklin, Brink & De Waal 2019; 2020), no field-based studies have been carried out to assess the ecological value of renosterveld buffers in terms of phytoremediation. The purpose of this chapter is to report on the determination of the potential of vegetative buffer strips in the Bot

River catchment (located in the Overberg, Western Cape, South Africa) to (phyto)remediate N and P pollution from agricultural sources before entry into stream and river channels. This involves an investigation of downslope changes to nutrient (N & P) concentrations in three study sites in the Bot River catchment to determine the effect that natural vegetation (renosterveld) fragments, where cultivated fields transition to natural vegetation, have on soil nutrient levels and the downslope movement of these nutrients. Thus changes in the nutrient contents of soils were investigated along hillslopes with an agricultural impacted site upslope and a reference renosterveld site downslope. If these buffer zones of natural vegetation do prove to provide important ecosystem services, then the conservation prioritisation of these zones should only increase.

6.2 ENVIRONMENTAL CONTEXT

The study site (Figure 6.2) is located in the Bot River catchment, which is home to a unique landscape of endangered Western Rûens Shale Renosterveld vegetation dominated by extensive drylands agriculture, particularly oilseed in addition to cereal crops. Shale renosterveld is the dominant veld type in the area, especially in the low-lying areas, while the slopes of the mountainous region to the west of the Bot River (and source of many of its tributaries) are covered largely by fynbos. Renosterveld is an evergreen shrubland or grassland principally covered by asteraceous shrubs with a grass understorey and an abundance of geophytes (Boucher 1980). Both the floral diversity and the number of geophytes increase with fire frequency and soil fertility (Kruger 1979; Rebelo et al. 2006). The lowland renosterveld in the area forms part of the Fynbos biome – a species-rich floral kingdom – but typically occurs on fine-grained, clay-rich soils as opposed to sandy, nutrient-poor soils on which fynbos is located (Rebelo et al. 2006). Significantly, seasonally-wet areas, streams and rivers act as crucial ecological corridors for renosterveld vegetation owing to the relatively lower levels of disturbance in these habitats.

Due to the high degree of degradation and fragmentation of renosterveld vegetation (Curtis, Stirton & Muasya 2013; Rebelo et al. 2006; Topp & Loos 2019), much of the remaining vegetation is located adjacent to watercourses, thus tracking river corridors through the landscape. These river corridors have great importance for biodiversity conservation (Curtis, 2016) and they act as a last line of defence for biodiversity under serious threat from landuse change and pollution (West, Cairns & Schultz 2016). Plant and animal communities are supported by rivers that link the major habitats of the Cape lowlands, help to disperse seeds and also act as important corridors for animal movement, particularly pollinators (Von Hase et al. 2003).

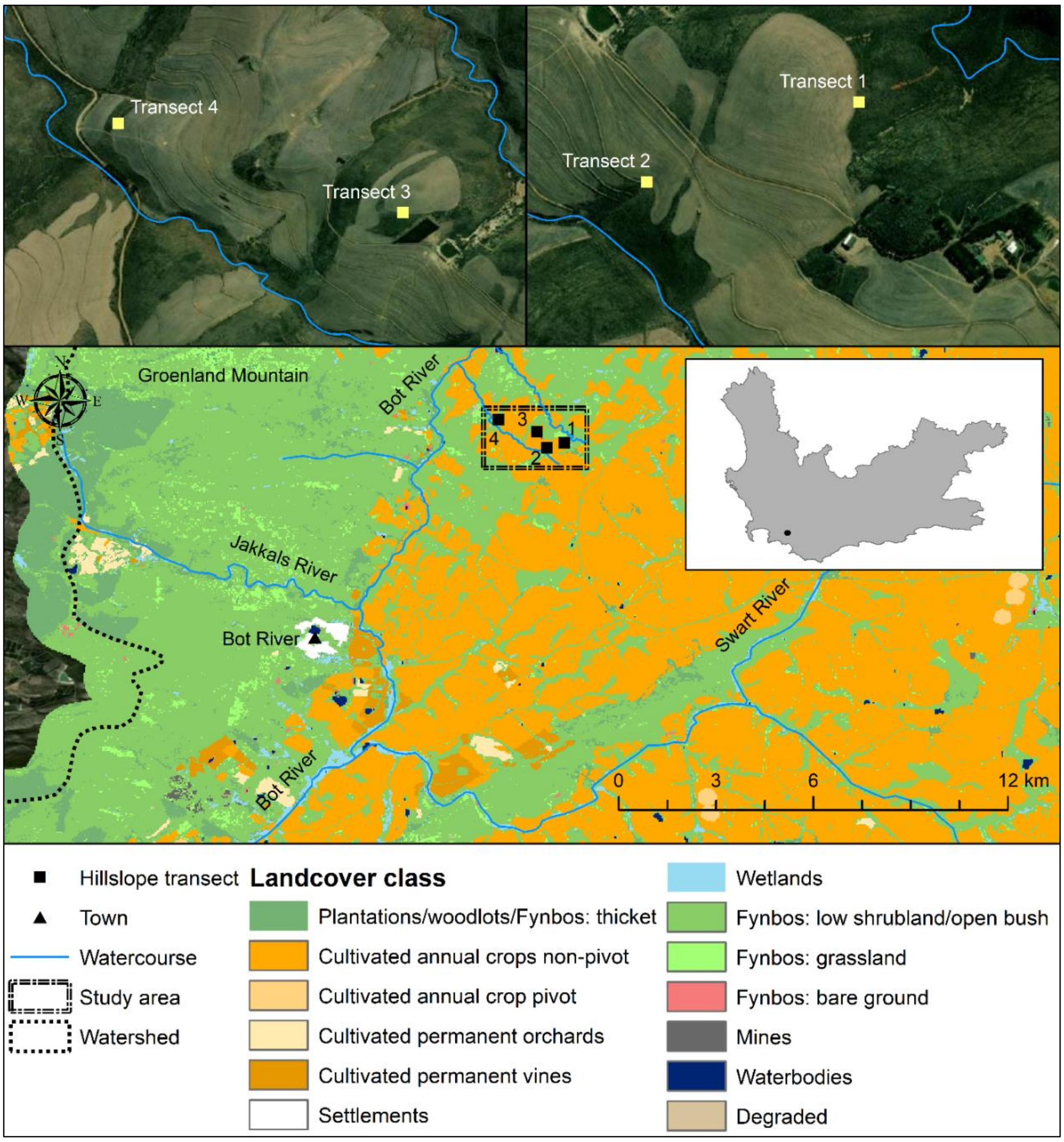


Figure 6.2 The Bot River catchment with four study sites (hillslope transects) and landuse indicated.

The region in which the study sites are located is characterised by winter rainfall (May to August) and dry summers (Bailey & Pitman 2015; Rebelo et al. 2006). Temperature ranges typically between 5°C (July average minimum) and 27°C (January average maximum) (Rebelo et al. 2006). The region is also host to lithic, duplex, plinthic, gleyic and cumulic soils within the various landtypes (Fey 2010). Shallow, stony lithic soils dominate to form convex slope crests and gentler footslopes that are common in the region (Fey 2010). The presence of duplex soils in the area indicates the enrichment of the subsoil with clay or a binary parent material (gravelly colluvium overlying residual shale). Thus there is a marked increase in the clay proportion in the B horizon when compared with the horizon above it (Fey 2010), resulting in the subsoil being a limiting factor to both root growth and the movement of water through the soil. As a result, overland flow contributions to streamflow may be far greater than the infiltration and throughflow of water in the soil, however, there may also be substantial shallow lateral soil water movement above the clay horizon (perched water table). In the Western Rûens Shale, clays and loams produced from the Bokkeveld Group shales are common – particularly the Glenrosa and Mispah forms (Fey 2010; Rebelo et al. 2006).

6.3 METHODS

A multimethod analysis of downslope changes in soil nutrient (P, NH_4^+ and NO_3^-) concentrations was performed in five stages. For Stage 1, three study sites (hillslope Transects 1 to 3) of approximately 40 m length and similar slopes (20-25%) were selected (Figure 6.2). These transects were selected on the basis of the nature of each hillslope where cultivated fields slope downwards to renosterveld fragments that act as buffers. A fourth transect (Transect 4) was added during the course of this study as described in Section 6.3.4. Samples were collected at four points down these transects at ~ 10 m intervals. The initial assessment of these sites was conducted by scanning for apparent electrical conductivity (EC_a) using a Geonics EM-38 MK2 scanner to provide context on whether any field boundary (the delineated line between agricultural fields and natural vegetation) differences in soil conductivity exist. During Stage 2, the soils of Transects 1 to 3 were described (Turner 1991) and classified using the taxonomic system for South Africa (Soil Classification Working Group 1991). In Stage 3 the particle size distribution of the soils was determined. In Stage 4 additional soil samples were collected for determination of phosphorus and nitrogen levels. Soil samples for this fourth stage were collected at three intervals, namely winter, spring and summer. Stage 5 involved adding a fourth transect (Transect 4) and evaluating the spatio-temporal variations in isotopic composition of N and C in the soils of each transect, as well as the concentrations of total N, P and C.

6.3.1 Stage 1: EM scan

During Stage 1 a Geonics EM-38 MK2 sensor was used for the characterisation of near-surface salinity. Electromagnetic (EM) induction scanners are used widely for the purposes of interpreting soil spatial variability. The scan was performed in May 2019. The MK2 sensor measures the electrical response of a soil and thus determines the apparent electrical conductivity (EC_a) of surface and near-surface soils. EC_a can be used as a proxy for selected soil properties, such as salinity (Gebbers et al. 2009; Van der Kroef et al. 2020). EC_a in soils is influenced by a number of factors, including salinity, texture, moisture and organic content (Corwin & Plant 2005; Shaner et al. 2008; Tsoulias, Gebbers & Zude-Sasse 2020). The scans were done while the soils were still dry (pre-rains) in the hope that the effects of soil moisture on EC_a measurements would be minimised. The EM38-MK2 enables measurements of EC_a in its quad-phase.

The EC_a data were obtained by operating the scanner in its shallow (0.5 m) and deep (1.0 m) settings in its horizontal and vertical modes. Three transects (1 to 3) were gridded by using tape measures and markers and then systematically scanned by transect walks. Transect lines were spaced two metres apart to ensure adequate coverage. The instrument was held approximately 10 cm above the ground surface during data collection and the scans covered an area of approximately 800 m^2 (40 m x 20 m) for each transect.

The EC_a values measured at 0.5-m and 1.0-m depths were spatially interpolated using the ordinary Kriging algorithm. This provided an estimated surface model from the spatial description of the scattered EC_a data points generated by the scanner. Kriging was performed using the package gstat (Pebesma 2004) in R. Any consideration of data recorded by the scanner assumes lateral homogeneity in soils, which, if not present, may result in anomalous measurements generated by the instrument (Bennett, George & Ryder 1995), particularly when soils contain large boulders near the surface. Thus, outliers in each transect (data below $Q_1 - 1.5 \times IQR$ or above $Q_3 + 1.5 \times IQR$) were removed (Upton & Cook 1996). To counter any measurement anisotropy bias, semi-variogram models were fitted in various directions and the best fitting model was selected.

6.3.2 Stage 2: Soil classification

For Stage 2 soil classification took place in May 2019. A tractor-loaded-backhoe (TLB) was used to excavate three observation pits at each hillslope (1 to 3) (Figure 6.3A). Two observation pits were dug in the cultivated field and one in the renosterveld for each hillslope. Soil profiles were first described by their slope position, parent material, diagnostic soil horizons, structure and mottling and then classified according to the Soil Classification Working Group (1991) classification system.

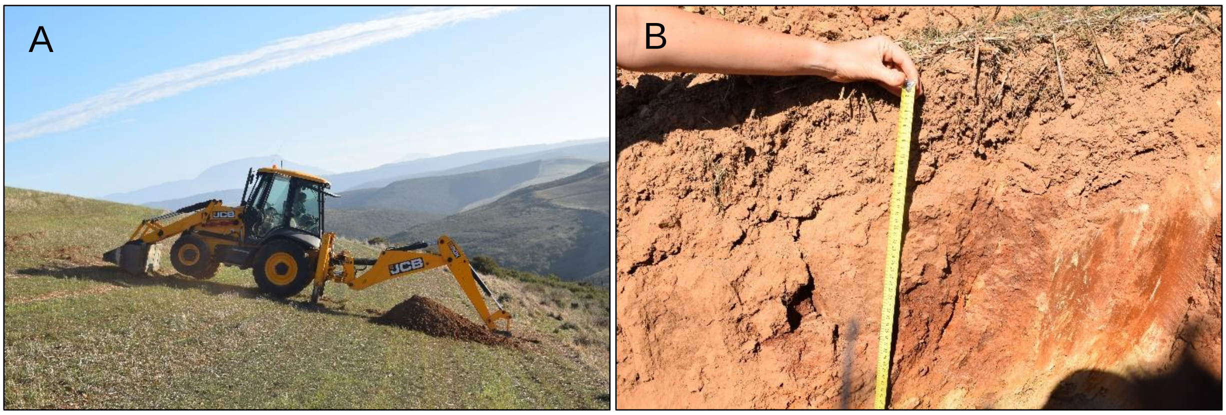


Figure 6.3 A tractor-loaded-backhoe digs an observation pit at Transect 1 (A) and horizon depths were determined (B).

Soil samples from each horizon were taken for laboratory analysis of particle size distribution (Stage 3).

6.3.3 Stage 3: Particle size distribution

The samples were air dried, weighed and crushed using a pestle and mortar before being passed through a 2 mm sieve to separate the coarse and fine fractions. Representative samples of the fine soils were weighed (20 g) and pretreated to remove organic matter by adding water and then 5 cm^3 H_2O_2 to the suspension. Excess water was evaporated, more H_2O_2 was added and this process repeated until all organic material was removed as indicated by bleached samples and lack of frothing. After final addition of H_2O_2 the samples were heated for one hour to remove remaining H_2O_2 (Soil Classification Working Group 1991). Soluble compounds were washed from the samples by centrifugation and soils were dried overnight. Samples were treated with 150 cm^3 citrate-bicarbonate buffer solution and 3 g $Na_2S_2O_4$ were added. The samples were heated for 30 minutes in a water bath at 80°C and centrifuged before being washed with 50 cm^3 distilled water. Calgon was added and the sand, silt and clay fractions determined using the methods described by the Soil Classification Working Group (1991). Sand fractions were separated by washing the sample through a 0.053-mm mesh sieve, drying the sand fraction (>0.053 mm) in an evaporating dish and transferring it to a nest of sieves (0.5 mm, 0.25 mm, 0.106 mm, 0.053 mm and a pan) before weighing each fraction. The separated silt and clay fractions were added to a 1-dm^3 cyclinder (which was then topped up to the 1-dm^3 mark) and allowed to rest until equilibration (at a constant room temperature of 20°C). Samples were then disturbed using a hand stirrer and a Lowy pipette was used to draw 25 cm^3 of each sample at 10-cm depth after 4 min 39 sec for fine silt (0.02 mm) and 7 hrs 45 min for clay (0.002 mm) fractions. These suspensions were evaporated and weighed.

6.3.4 Stages 4 and 5: Sample collection

For Stage 4 the samples for the determination of soil nutrient levels were collected over a seven-month period (2019-2020). Plant-available (bio-available) phosphorus as well as nitrate and ammonia levels in the soil were quantified. Samples were collected at four different points down each site’s hillslope during winter (August 2019), spring (November 2019) and summer (February 2020) using a manual auger. These points were located adjacent to the location of each observation pit dug for soil classification. An additional sample was collected further downslope in the natural vegetation buffer strips. Samples were taken from predominantly A and B horizons at each point located above the layers confining subsurface flow and in the root zone of crops and natural vegetation (see Table 6.2). Samples were taken in triplicate at each site.

During Stage 5 samples were collected on three days: (1) prior to fertiliser application, (2) shortly after application and (3) a month thereafter in 2021. A kernel-based NPK fertiliser that dissolves in the soil during rain events was applied simultaneously with sowed seeds in the cultivated fields of each hillslope. The fertiliser product, produced and distributed by Omnia Nutriology (Omnia Holdings Limited 2022), has the specific elemental components listed in Table 6.1. However, in the case of Transect 1 the local farmer had decided to leave the land fallow for 2021, thus no fertiliser was applied to the field. Consequently, a fourth site was selected to replace study Site 1. For Stage 5 total P and total N in the soils were determined while nitrogen and carbon isotopic analysis was conducted on the samples to determine the sources of these nutrients (natural or anthropogenic).

Table 6.1 Composition of Omnia fertiliser applied to agricultural fields during this study.

Fertiliser component	Concentration (g/kg)
N	257
P	66.0
K	36.7
B	0.40
S	44.0
Cu	0.70
Mn	1.50
Zn	1.50

Sowing and fertilisation occurred in May 2021. Samples were collected from the A horizon (20-cm depth) on 14 May (preceding fertilisation), 28 May (post-fertilisation) and 28 June 2021 using a manual auger. Samples were placed in labelled plastic bags before being transported to the laboratory for drying and further analysis.

6.3.5 Determination of soil phosphorus levels

Stage 4 of this study measured plant-available phosphorus (Section 6.3.5.1) while Stage 5 involved determination of total P in the soils (Section 6.3.5.2).

6.3.5.1 Plant-available phosphorus: 2019-2020 (Stage 4) analysis

The Bray II method (Bray & Kurtz 1945) was used for the determination of plant-available phosphorus. This method extracts acid soluble and available or reserve phosphates present in the soil (White 2019). Bray II is appropriate for acidic soils (Olsen & Sommers 1982; Pierzynski 2000) and is commonly used in South African analytical laboratories.

Soil samples were air dried (25 to 30°C) and crushed to pass through a 2 mm sieve (Pierzynski 2000). The fine fraction was analysed for available P. Bray II (0.1M HCl + 0.03 M NH_4F) extract was used with molybdate blue for colorimetric determination of plant-available phosphorus. A soil to solution ratio of 1:7 was used for extraction from the soil samples with an extraction time of 40 sec. Standard solutions were used to create a calibration curve for P determination. Sample absorbance was measured at 660 nm (Knowles & Plaxton 2013; Olsen & Sommers 1982). Measured values in ppm were converted to mg/kg of soil.

6.3.5.2 Total phosphorus: 2021 (Stage 5) analysis

The analysis for total phosphorus took place at Stellenbosch University's Central Analytical Facility laboratories in the ICP-MS & XRF unit. Analysis was conducted by using inductively coupled plasma optical emission spectroscopy (ICP/OES). Samples were dried in an oven at 60°C for three days and were subsequently milled with a mortar and pestle and sieved through a 1 mm mesh to separate stones from the soil used for analysis. Samples were weighed to a 0.5-g homogenous sample. Samples were passed through a nebuliser that creates a fine aerosol. The Thermo iCAP 6000 nebuliser is used when samples range from high mid ppb to high ppm levels. Total P was recorded as mg P/kg.

6.3.6 Determination of soil nitrogen levels

As with the determination of P, Stage 4 of this study measured NO_3^- and NH_4^+ (Section 6.3.6.1) while Stage 5 involved measuring of total N and C in the soils (Section 6.3.6.2).

6.3.6.1 Nitrate and ammonia concentrations – 2019-2020 (stage 4) analysis

A 2M KCl extract was used on soil samples with a 1:10 soil to solution ratio (Mulvaney 1996). Measurements of ammonia and nitrate levels were taken using a photometer combined with reagent kits (Spectroquant® ammonia and nitrate tests from Merck) (e.g. Martínez and Dussán 2018). The Merck ammonia Spectroquant® test kits have a measuring range of 2.0 to 150 mg/l NH_4-N (absorbance was measured at 667 nm), while the nitrate test has a measuring range of 0.2 to 20.0 mg/l NO_3-N (absorbance measured at 540 nm) where the suggested wavelengths for measurement were recommended by Mulvaney (1996). Standard solutions were used to create a calibration curve for NH_4-N and NO_3-N determination. Finally, measured values in ppm were converted to mg/kg of soil.

6.3.6.2 Total nitrogen and carbon – 2021 (stage 5) analysis

For total nitrogen and carbon, analysis was conducted using an Elementar Vario El Cube Elemental Analyser (Russow & Goetz 1998; Schulz, Härtling & Stange 2011). A standardised mass of each sample was placed into an aluminium receptacle with 5 mg of tungsten trioxide (WO_3) powder added. Samples were heated in a combustion column at a temperature of 1050°C and dosed with O_2 which caused the C and N in each sample to combust and generate reaction products: CO_2, H_2O, N_2, NO_x, SO_2 and SO_3. Adsorption columns were then heated sequentially to allow desorption of different products which were then carried through to the thermal conductivity detector (TCD). All samples were analysed using a certified reference material to demonstrate method accuracy. Final carbon and nitrogen levels are given as a mass percentage.

6.3.7 Carbon and nitrogen isotopic analysis

For the analysis of isotopes samples were weighed into tin cups to an accuracy of one microgram on a Sartorius M2P microbalance. Samples were combusted in a Flash 2000 organic elemental analyser and the gases passed to a Delta V Plus isotope ratio mass spectrometer (IRMS) using a Thermo Scientific Conflo IV gas control unit. In-house standards were calibrated against International Atomic Energy Agency (IAEA) standards. Nitrogen is expressed in terms of its value relative to atmospheric nitrogen, while carbon is expressed in terms of its value relative to Pee-Dee Belemnite (Brand et al. 2014; Craig 1957). Accuracy was determined using the standard deviation of repeat runs, the allowable levels being set to <0.2‰ for nitrogen and <0.3‰ for carbon.

6.3.8 Statistical treatment of data

For the comparison of sample means between cultivated fields and renosterveld soils Welch two sample t-tests (Welch 1938; Yuen 1974) were used at the 95% significance level based on triplicate samples. Data were first displayed graphically to evaluate whether they followed a Gaussian distribution and then treated statistically using R. Graphics were generated using ggplot and pairwise comparisons are presented by letter display (Piepho 2018).

6.4 RESULTS

An overview of the results for each sample site is given in Table 6.2. Each site’s field position, soil form and texture class are described, coupled with the mean nitrate, ammonium, phosphorus and carbon concentrations tested during the different stages of sample collection and analysis.

Table 6.2 Summary of results reflecting soil characteristics and nutrient concentrations for each sample site with colour scale indicating total N, P and C magnitude from green (low) to red (high).

Transect	Site number	Field position	Soil form and family	Texture class (A horizon)	2019 analysis Mean NO_3 (mg/kg)	2019 analysis Mean NO_4 (mg/kg)	2019 analysis Mean bio-available P (mg/kg)	2021 analysis Total N (%)	2021 analysis Total P (mg/kg)	2021 analysis Total C (%)
1	1	Cultivated upslope	Ss2200	Clay loam	53.8	64.1	39.7	0.21	551	2.15
	2	Cultivated downslope	Sw1/2111	Clay loam	62.7	44.8	29.9	0.22	717	2.24
	3	Renosterveld upslope	Sw2110	Clay loam	82.3	31.0	4.20	0.23	510	2.75
	4	Renosterveld downslope	-	-	66.6	38.6	1.20	0.20	419	2.39
2	1	Cultivated upslope	Km1120	Clay loam	70.2	36.5	50.4	0.15	419	1.58
	2	Cultivated downslope	Km111/20	Silt loam	76.1	51.8	38.9	0.12	406	1.24
	3	Renosterveld upslope	Km1120	Clay loam	69.3	34.5	9.80	0.26	498	2.95
	4	Renosterveld downslope	-	-	66.3	24.3	8.00	0.16	381	1.75
3	1	Cultivated upslope	Ss2100	Loam	97.5	14.1	71.6	0.18	674	2.02
	2	Cultivated downslope	Tu2120	Clay loam	94.0	31.4	67.4	0.21	688	2.31
	3	Renosterveld upslope	Sw2121	Loam	61.6	38.0	10.8	0.27	466	3.33
	4	Renosterveld downslope	-	-	54.7	26.6	4.90	0.17	379	2.17
4	1	Cultivated upslope	-	-	-	-	-	0.16	567	1.65
	2	Cultivated downslope	-	-	-	-	-	0.15	450	1.66
	3	Renosterveld upslope	-	-	-	-	-	0.16	287	1.85
	4	Renosterveld downslope	-	-	-	-	-	0.16	262	2.02

More detailed accounts of the results are reported next for each research stage. First, the study site and soil characteristics regarding EC_a (Section 6.4.1), soil horizon depth and soil form (Section 6.4.2), particle size distribution where confining layers to subsurface flow were identified (Section 6.4.3), soil nutrients (C, P, N, NH_4^+ and NO_3^-) (Section 6.4.4 and 6.4.5) and the isotopic composition of N and C in the soil samples (6.4.6).

6.4.1 Apparent electrical conductivity

The spatial interpolations of EC_a measurements for each site (transect) at two depths are given in Figures 6.4, 6.5 and 6.6. For Transect 1 (Figure 6.4), EC_a values at 1.0-m depth are significantly ($P \leq 0.05$) lower (mean = 13.7 mS/m) than those at 0.5 m (mean = 24.2 mS/m). A strong field boundary effect is evident where a significant difference in EC_a at 0.5-m and 1.0-m depths exists where higher EC_a values are present in cultivated fields and lower values in the natural vegetation area. A rock bank is present to the west of the field boundary (probably impeding ploughing). EC_a values at 0.5 m ranged between 1 mS/m and 52 mS/m, while values at 1.0-m depth lay between 0 mS/m and 34 mS/m.

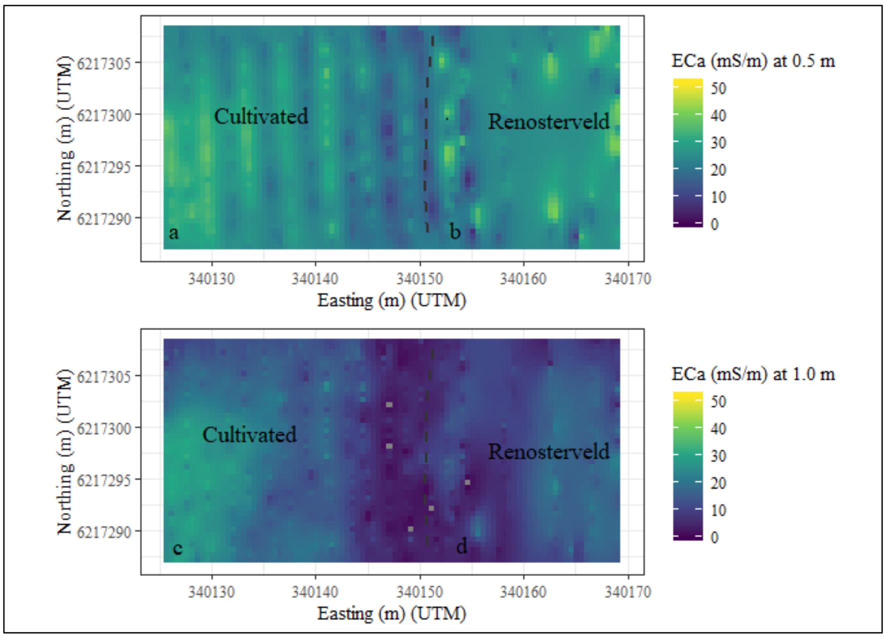


Figure 6.4 EC_a for Transect 1 - slope from left to right with field boundary line indicated. Mean values with the same letters are not significantly different at P ≤ 0.05. Means are the average values of the data from 2406 data points.

As with Transect 1, EC_a for Transect 2 is significantly lower at 1.0 m (mean = 4.13 mS/m) than at 0.5 m (mean = 23.1 mS/m) (Figure 6.5). There is a statistically significant field boundary effect evident at 0.5 m and 1.0 m, where another rock bank exists. EC_a values at 0.5 m ranged between 3 mS/m and 43 mS/m, while at 1.0-m depth values ranged between 0 mS/m and 16 mS/m.

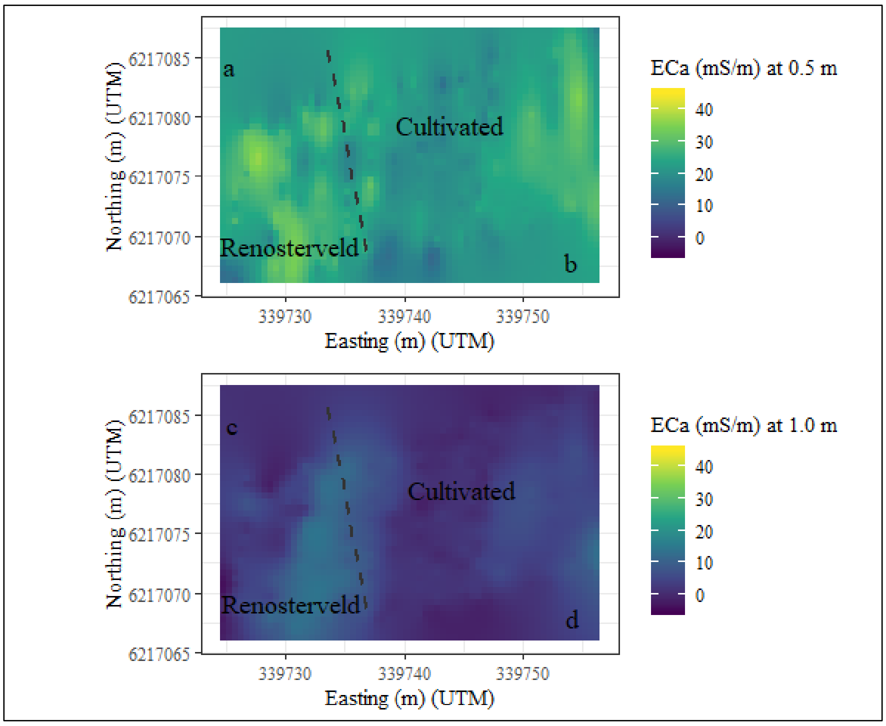


Figure 6.5 EC_a for Transect 2 - slope from right to left with field boundary line indicated. Mean values with the same letters are not significantly different at P ≤ 0.05. Means are the average values of the data from 2304 data points.

For Transect 3, EC_a ranges between 0 mS/m and 36 mS/m at 0.5 m (mean = 17.5 mS/m), with greater variability between 0 mS/m and 45 mS/m at 1.0 m (mean = 19.4 mS/m) (Figure 6.6). This differs from the results of Transects 1 and 2, however there is a statistically significant field boundary effect where EC_a changes between cultivated fields and natural vegetation at the field boundary (line) at both 0.5 m and 1.0 m.

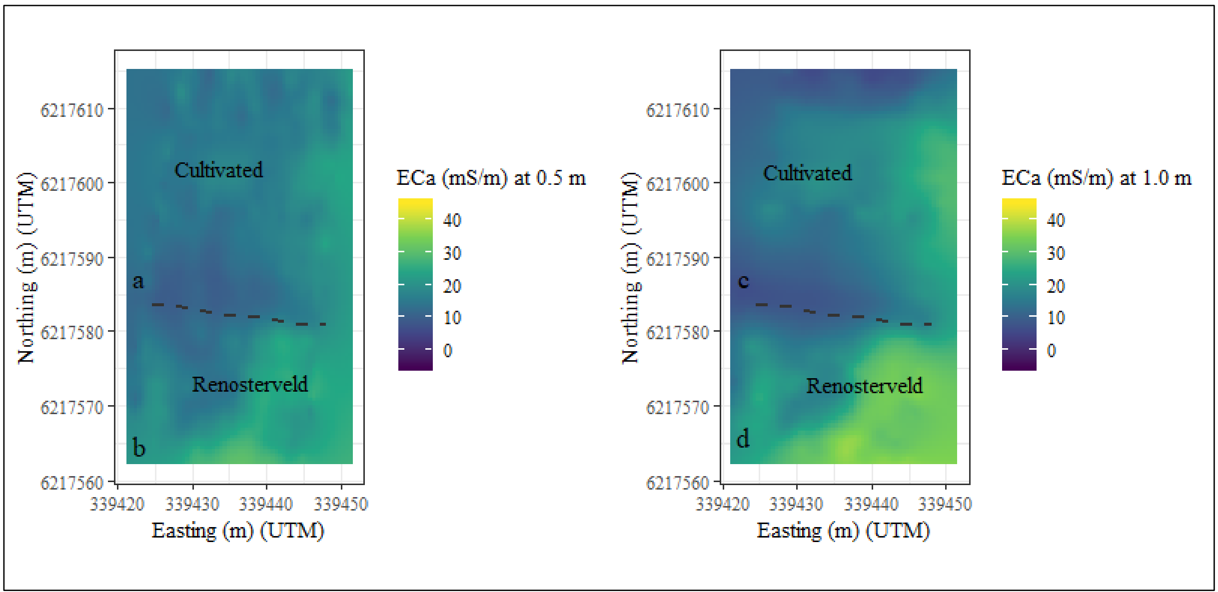


Figure 6.6 EC_a for Transect 3 - slope from top to bottom with field boundary line indicated. Mean values with the same letters are not significantly different at $P \leq 0.05$. Means are the average values of the data from 4197 data points.

EC_a is typically lower at greater depths (1.0 m) for all three transects than at a shallower level (0.5 m), while in all three transects a field boundary effect is evident. A clear field boundary effect exists (i.e. a marked change in EC_a between the cultivated field and natural vegetation at the field boundary line) in all three sites, particularly at 1.0-m depth, although this is also present at 0.5 m for Transects 2 and 3. Higher EC_a values are seen more in natural vegetation soils than in the cultivated fields. This indicates that a change in soil texture, salinity or moisture content is evident in each site, indicating changing soil depth and the presence of underlying rock banks in the natural vegetation zones or slope effects.

6.4.2 Soil classification

The classification of the soil in each pit is listed in Table 6.3. At the cultivated upslope pit of Transect 1 (1.1), a Sterkspruit (Ss) form with a bleached A horizon and a red prismacutanic[18] B horizon is present. A Swartland (Sw) form with a bleached A horizon and non-calcareous pedocutanic[19] B and upper C horizons was observed at the cultivated field downslope pit, while a Sw form with a bleached A horizon and subangular B horizon was noted at the renosterveld pit. Transect 2 differed from the other two transects in that an E horizon was present. Here all three observation sites had Klapmuts (Km) form soils with a grey E horizon, a non-red, medium-coarse, angular pedocutanic B horizon. Transect 3 had more variation in soil forms down the slope profile where the cultivated upslope site had an Ss form with a bleached A horizon and a non-red prismacutanic B horizon. The second site – cultivated, downslope – displayed a Tukulu (Tu) form with a non-red, luvic, neocutanic B horizon and the third site in the natural vegetation was an Ss form with a bleached A horizon and a non-red, prismacutanic B horizon.

[18] "A horizon with an abrupt transition with an overlying A [or E] horizon with respect to texture, structure or consistence; the structure is strong prismatic or columnar." (Van der Walt & Van Rooyen 1995, p.47)

[19] "A horizon with strong blocky structure and clearly expressed cutans." (Van der Walt & Van Rooyen 1995, p.47)

Table 6.3 Classification of soils in the study area.

Transect	Site	Horizon	Diagnostic horizons	Lower depth (mm)	Colour (Munsell code)	Soil form	Soil group (Fey 2010)
1	Cultivated upslope (1.1)	A	Orthic A	200	7.5YR6/4	Ss 2200	Duplex*
		B	Prismacutanic B	500	7.5YR4/6		
		C	Lithocutanic B (Saprolite)	1500+			
			Saprolite				
	Cultivated downslope (1.2)	A	Orthic A	150	7.5YR6/4	Sw 1/2111	Duplex*
		B	Pedocutanic B (Neocutanic B)	500	7.5YR5/4		
		C	Lithocutanic B (Saprolite)	900+			
			Saprolite				
	Renosterveld (1.3)	A	Orthic A	100	7.5YR6/4	Sw 2111	Duplex*
		B1	Pedocutanic B	400	7.5YR5/4		
		B2	Lithocutanic B	900+	7.5YR4/6		
		C	Saprolite				
2	Cultivated upslope (2.1)	A	Orthic A	200	10YR7/2	Km 1120	Duplex*
		E	E horizon	400	10YR7/2		
		B	Pedocutanic B	550	10YR4/6		
		C	Saprolite	550+			
	Cultivated downslope (2.2)	A	Orthic A	150	10YR7/2	Km 111/20	Duplex*
		E	E horizon	450	10YR7/3		
		B	Pedocutanic B	900	10YR5/6		
		C	Saprolite	900+			
	Renosterveld (2.3)	A	Orthic A	100	10YR7/3	Km 1120	Duplex*
		E	E horizon	350	10YR7/3		
		B2	Pedocutanic B	600	10YR6/8		
		C	Saprolite	600+			
3	Cultivated upslope (3.1)	A	Orthic A	200	10YR6/2	Ss 2100	Duplex*
		B	Prismacutanic B	400	10YR3/2		
		C	Saprolite	400+			
	Cultivated downslope (3.2)	A	Orthic A	100	10YR7/2	Tu 2120 (Km)	Cumulic/ Duplex*
		B1	Neocutanic B/E horizon	300	10YR6/2		
		B2	Pedocutanic B	500	10YR3/3		
		C	Saprolite	500+			
	Renosterveld (3.3)	A	Orthic A	300	10YR6/2	Sw 2121	Duplex*
		B	Pedocutanic B	400	10YR4/2		
		C	Saprolite	400+			

*** WRB classification (Food and Agriculture Organization 2015) – Luvisol**

These soils are all largely duplex soils that have a clay-enriched B horizon. Notably, most of the soil forms are either Ss, Sw or Tu with bleached A horizons or Km (Transect 2) with a grey E horizon and bleached B horizon, indicating water movement through these horizons.

6.4.3 Particle size distribution

Transect 1 showed varying soil textures alternating between clays, clay loams and silty clay loam (Table 6.4). There are clear horizons that confine subsurface flow in each observation site where the clay fractions are substantially higher than those of the horizons above. For Site 1.1 (i.e. Transect 1, cultivated upslope), the B horizon is the confining layer as the clay fraction is >50%, while subsurface flow at Site 1.2 is limited by the C horizon and at Site 1.3 by the B horizon. In Transect 2, texture classes range between clays, silt loams and clay loams, while the B horizon impedes vertical water movement as the high clay percentages (46-56%) when compared to the A and E horizons result in impervious zones. At Transect 3, soil texture classes range from clays and clay loams to loam. Again, the B horizon with clay fractions of approximately 45% is confining subsurface flow.

Table 6.4 Particle size distribution of soil horizons in the study area.

Transect/ Hillslope	Site	Horizon	Sand fraction (%)	Coarse silt fraction (%)	Fine silt fraction (%)	Clay fraction (%)	Soil texture class
1	Cultivated upslope (1.1)	A	23.3	21.6	20.4	34.7	Clay loam
1	Cultivated upslope (1.1)	B	3.1	14.8	31.0	51.1	Silty clay
1	Cultivated upslope (1.1)	C	49.1	9.5	20.4	21.0	Loam
1	Cultivated downslope (1.2)	A	26.4	27.0	10.0	36.6	Clay loam
1	Cultivated downslope (1.2)	B	17.8	16.0	30.5	35.7	Silty clay loam
1	Cultivated downslope (1.2)	C	9.8	15.4	20.6	54.2	Clay
1	Renosterveld (1.3)	A	28.5	15.5	20.1	35.9	Clay loam
1	Renosterveld (1.3)	B1	15.3	15.2	20.3	49.2	Clay
1	Renosterveld (1.3)	B2	5.5	13.0	20.8	60.7	Clay
2	Cultivated upslope (2.1)	A	21.4	21.1	20.3	37.2	Clay loam
2	Cultivated upslope (2.1)	E	26.5	29.0	20.4	24.0	Silt loam
2	Cultivated upslope (2.1)	B	12.6	9.6	30.9	46.8	Silty clay
2	Cultivated downslope (2.2)	A	24.0	20.9	30.7	24.3	Silt loam
2	Cultivated downslope (2.2)	E	24.4	19.9	9.9	45.8	Clay
2	Cultivated downslope (2.2)	B	8.7	15.0	20.5	55.8	Clay
2	Renosterveld (2.3)	A	23.8	20.9	20.3	35.0	Clay loam
2	Renosterveld (2.3)	B1	19.4	13.8	30.7	36.1	Silty clay loam
2	Renosterveld (2.3)	B2	19.5	13.2	20.4	46.9	Clay
3	Cultivated upslope (3.1)	A	26.0	18.5	31.0	24.5	Loam
3	Cultivated upslope (3.1)	B	22.6	2.3	30.8	44.3	Clay
3	Cultivated downslope (3.2)	A	29.3	13.5	20.3	36.8	Clay loam
3	Cultivated downslope (3.2)	B1	25.2	20.3	20.2	34.4	Clay loam
3	Cultivated downslope (3.2)	B2	11.6	12.3	30.8	45.3	Silty clay
3	Renosterveld (3.3)	A	26.5	26.3	20.8	26.4	Loam
3	Renosterveld (3.3)	B	20.4	15.2	20.4	44.0	Clay

The majority of soils in the study area have high clay content, especially in the B horizons characteristic of duplex soils. There was no marked difference in soil texture classes between cultivated field and renosterveld soil samples.

6.4.4 Phosphorus concentrations

This section describes the phosphorus content in the soils determined by two sets of analysis. First, plant-available phosphorus concentrations for soil samples taken in 2019 and 2020 were determined (Section 6.4.4.1) and then total P concentrations were measured for soil samples taken in 2021 (Section 6.4.4.2).

6.4.4.1 Plant-available phosphorus (2019-2020)

Changing plant-available P levels down the three hillslope transects (Transects 1 to 3) are depicted in Figure 6.7. For each transect depicted in Figure 5, the four sample positions down the hillslope are indicated and three samples were taken at each point to ensure accuracy of results. The location of field boundaries are indicated to show the differences in concentration levels between cultivated and renosterveld sites.

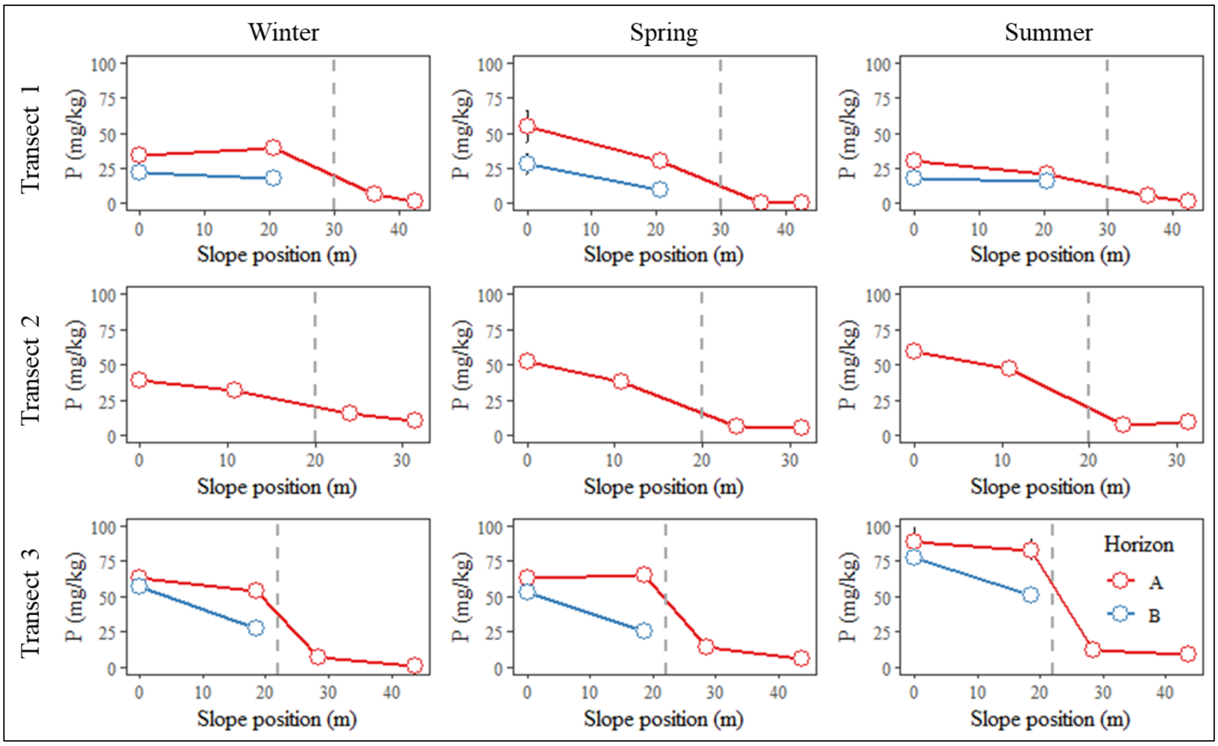


Figure 6.7 Mean plant-available phosphorus (P) concentrations based on three samples per site on slope transects in three seasons with field boundaries indicated.

There is some seasonal difference in the P concentrations in the soil samples but no clear pattern is apparent. Where a B horizon exists, P concentrations are higher in the A horizon than in the B horizon below, indicating limited vertical mobility of P in the soil. For all the transects, P levels in the renosterveld are substantially lower than those in the cultivated fields. Cultivated field concentrations are mostly within a range of 25-50 mg P/kg with a few exceptions (notably Transect 1 in Spring). P levels in the natural vegetation are lower, ranging between undetected to roughly 20 mg P/kg.

For all transects there is a difference between plant-available P levels in cultivated fields when compared with those in the natural vegetation. A marked decrease in P concentrations in the natural vegetation buffers is visible for all sites and all seasons (Figure 6.8).

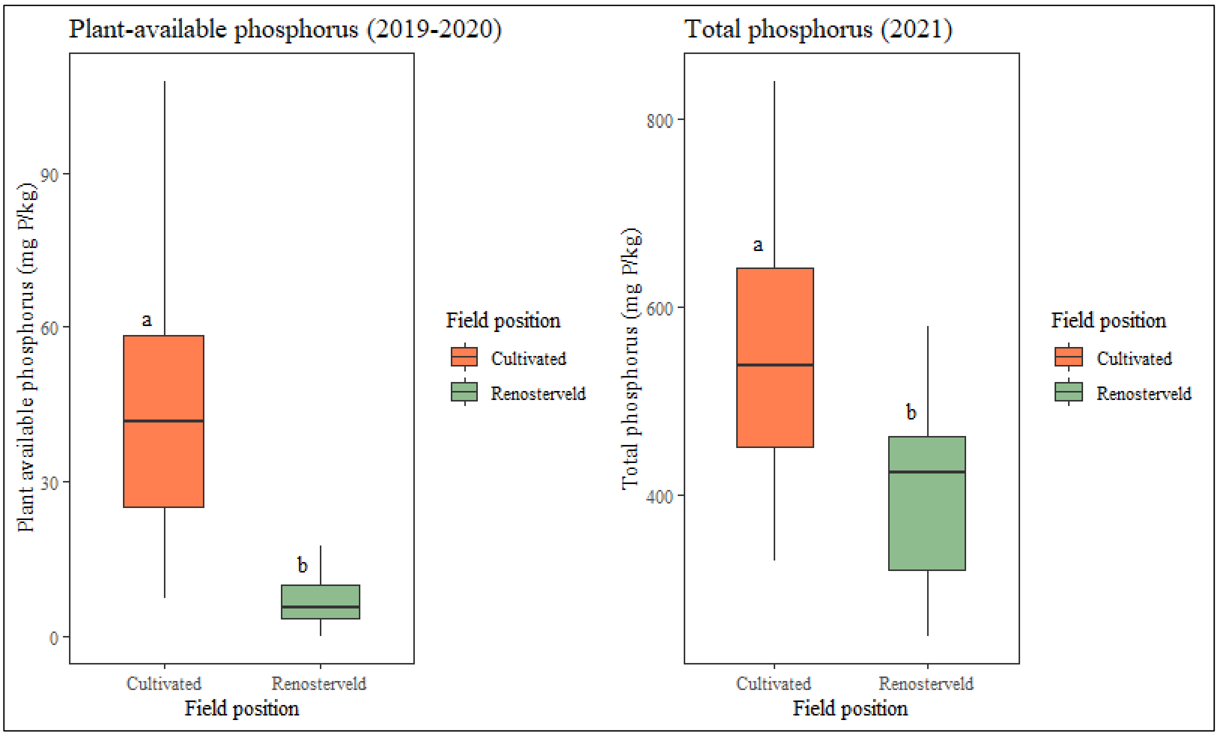


Figure 6.8 Boxplot comparing plant-available phosphorus (2019-2020) and total phosphorus (2021) concentrations in cultivated fields and renosterveld vegetation surrounds across all transects. Median values with the same letters for each analysis are not significantly different, while those with different letters were significantly different at $P \leq 0.05$.

The mean cultivated field plant-available phosphorus concentration across all seasons was 43.13 mg P/kg, which is significantly ($P \leq 0.05$) higher than that of renosterveld soils (6.48 mg P/kg). Plant-available phosphorus and total P concentrations in renosterveld soils are significantly lower than those found in cultivated field samples (Figure 6.8).

6.4.4.2 Total phosphorus in soils (2021)

The analysis of phosphorus samples taken in 2021 shows a similar pattern to those collected in 2019-2020 except for Transect 2. Total phosphorus (Figures 6.8 and 6.9) concentrations are significantly higher in the cultivated fields than in the renosterveld vegetation further downslope.

The pre-fertilisation samples (14 May 2021) generally showed lower total P concentrations than samples collected post-fertilisation (28 May and 28 June 2021).

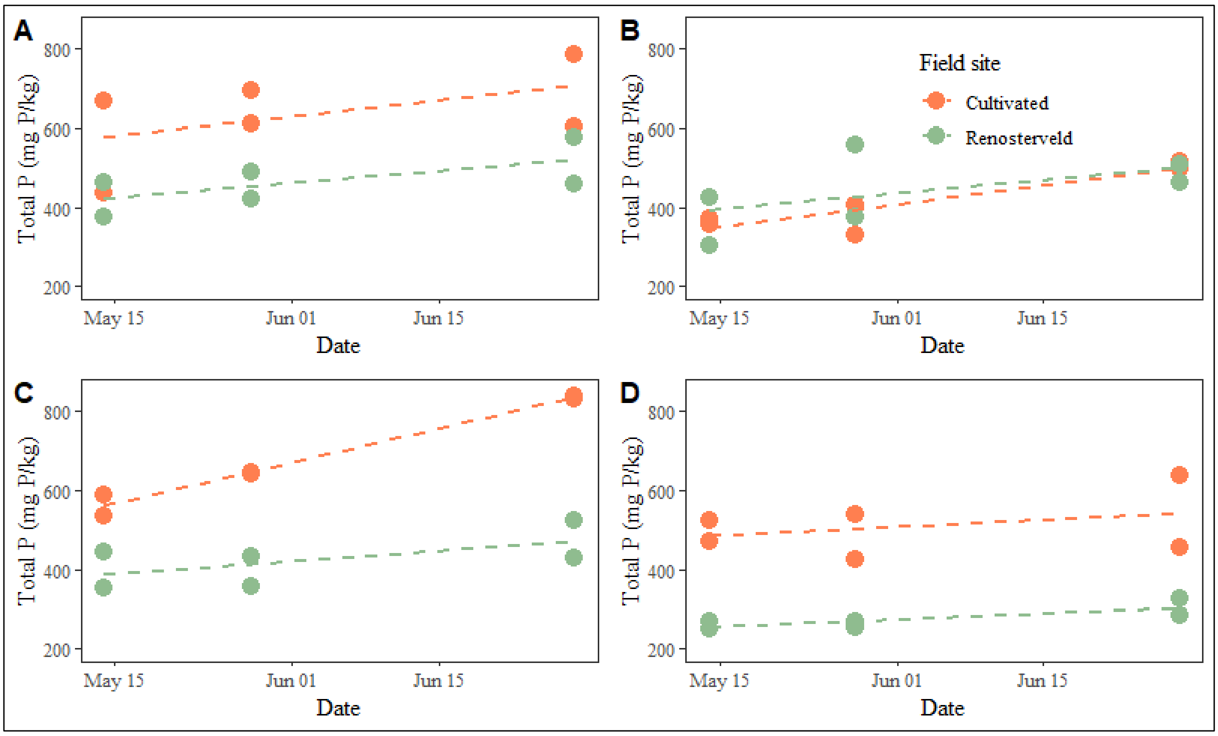


Figure 6.9 Variations in total phosphorus (P) for Transects 1(A), 2(B), 3(C) and 4(D) over time.

Soil phosphorus increased in both cultivated and renosterveld soils post-fertilisation, indicating movement of P downslope from fertilised fields. It is noteworthy that this pattern also exists for Transect 1 that was left fallow and unfertilised in 2021. Total phosphorus concentrations range between 249 mg P/kg and 842 mg P/kg of soil.

6.4.5 Nitrogen concentrations

This section describes the nitrogen content in the soils determined by two sets of analysis. First, the NH_4^+ and NO_3^- contents of soil samples taken in 2019 and 2020 were determined (Section 6.5.4.1) and then total N and C concentrations were measured for soil samples taken in 2021 (Section 6.5.4.2).

6.4.5.1 Ammonium and nitrate analysis in soils (2019-2020)

The concentrations of soil NH_4^+ for each sample are displayed in Figure 6.10. NH_4^+ levels range between undetected and in excess of 150 mg/kg.

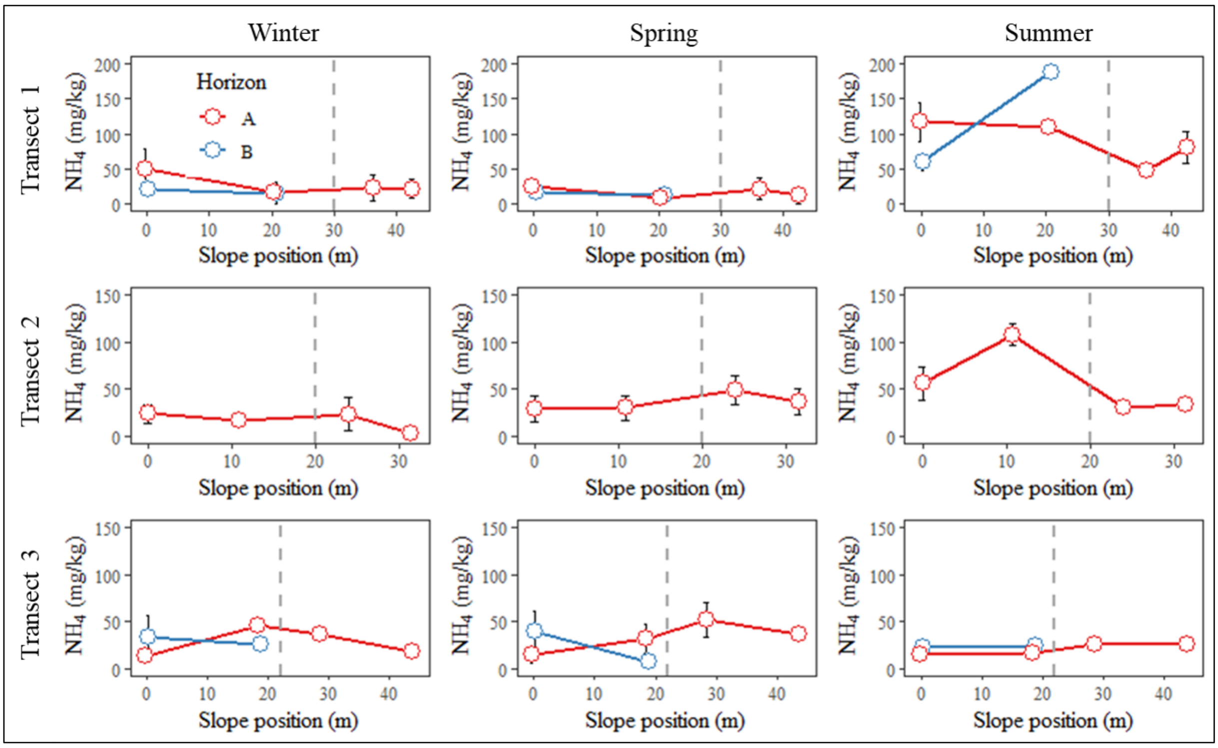


Figure 6.10 Mean ammonium (NH_4^+) concentrations based on three samples per site on slope transects in three seasons with field boundary indicated.

The summer NH_4^+ levels in Transect 1 are higher than those of winter and spring. It was observed that a flock of sheep had been grazing in the area prior to collection of the soil samples and could have influenced these concentrations. However, the sheep were not grazing in the natural vegetation which still displays high levels of NH_4^+. There is no difference between concentrations in the A and B horizons. An evaluation of the differences in NH_4^+ concentrations between cultivated fields and the surrounding renosterveld is shown in Figure 6.12, which show that no statistically significant difference in NH_4^+ concentrations between these landuses exists. Soil NO_3^- concentrations are exhibited in Figure 6.11.

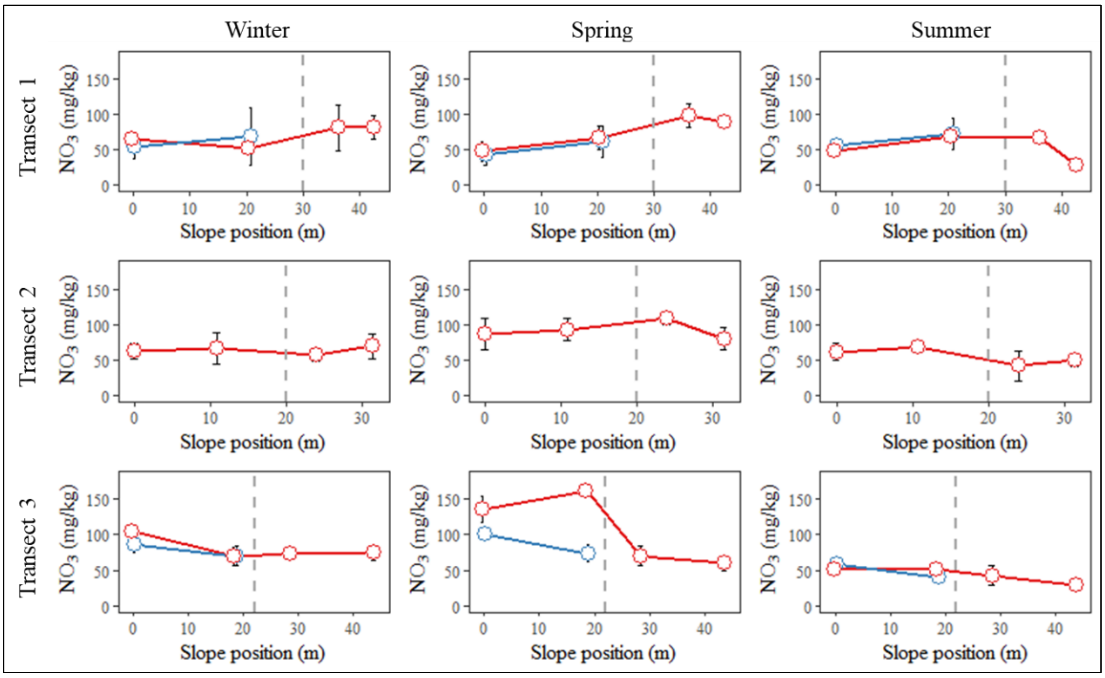


Figure 6.11 Mean nitrate (NO_3^-) concentrations based on three samples per site on slope transects over varying seasons with field boundary indicated.

Soil NO_3^- concentrations range between undetected and 150 mg/kg. Nitrate concentrations in the natural vegetation are comparable to those in cultivated fields (Figure 6.11), indicating some lateral movement of NO_3^- downslope from cultivated fields. Seasonal concentrations are again variable with no discernible seasonal pattern. There is also no significant difference between cultivated field and renosterveld NO_3^- soil samples (Figure 6.12).

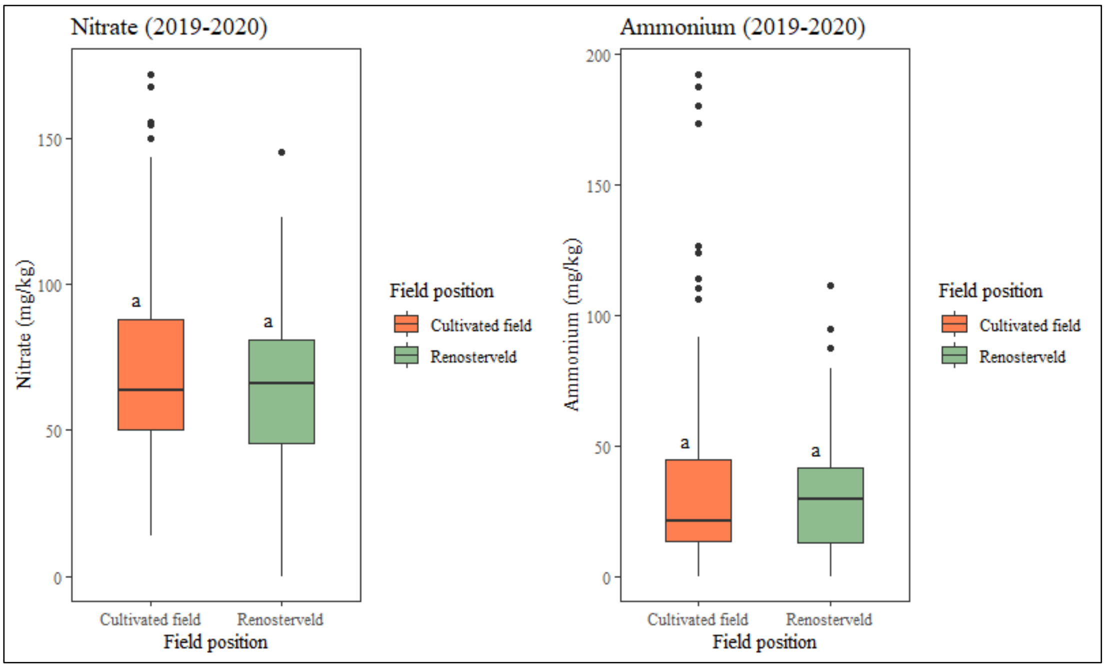


Figure 6.12 Boxplot comparing differences in nitrate and ammonium concentrations in the soils collected in cultivated fields and surrounding renosterveld vegetation. Median values with the same letters are not significantly different at $P \leq 0.05$.

Soil nitrate levels were higher than ammonium concentrations for cultivated fields and renosterveld soils (Figure 6.12). Notably, there were several outliers for ammonium concentrations in the cultivated field samples.

6.4.5.2 Analysis of total nitrogen and carbon in soils (2021)

Total N is higher in post-fertilisation samples of the cultivated fields (28 May and 28 June 2021) than in pre-field fertilisation samples (Figure 6.13) indicating the impact of NPK fertiliser application to these cultivated fields. The increase in nitrogen in renosterveld fragments indicates that nitrogen mobility from agricultural fields does take place after fertilisation.

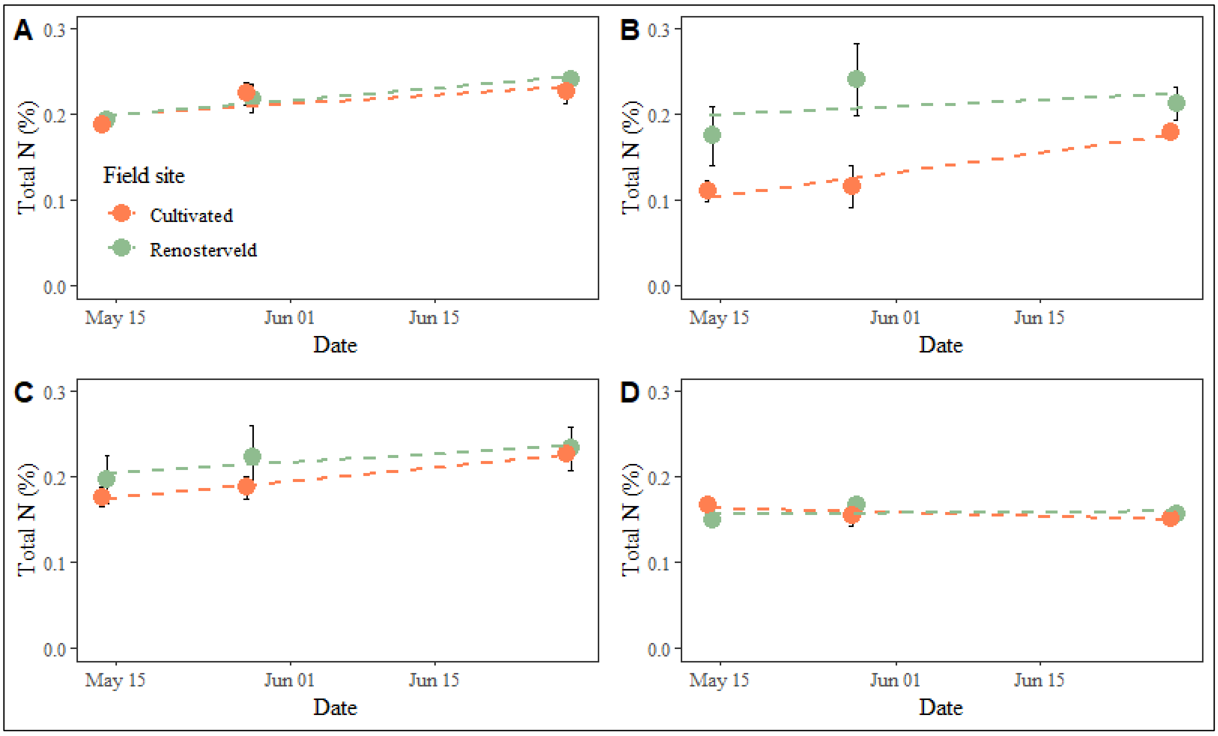


Figure 6.13 Variations in total nitrogen (N) for Transects 1(A), 2(B), 3(C) and 4(D) before and after fertilisation.

There is a limited temporal pattern of total carbon in the soil samples with some results indicating lower total carbon after fertilisation than existed before (Figure 6.14D). However, a significant difference between cultivated fields and renosterveld vegetation is evident.

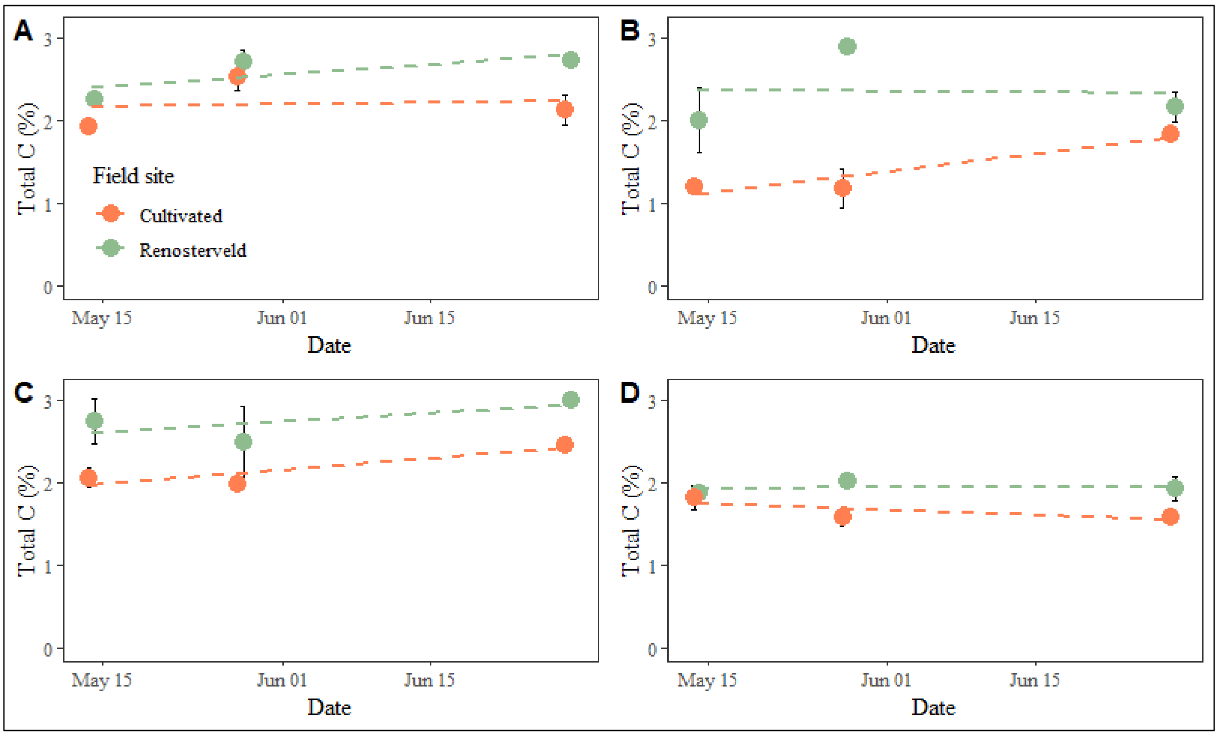


Figure 6.14 Variations in total carbon (C) for Transects 1(A), 2(B), 3(C) and 4(D) before and after fertilisation.

Total carbon in the naturally vegetated hillslope sections is significantly higher ($P \leq 0.05$) than total carbon in cultivated fields (Figure 6.15). Median total carbon is 0.5% greater in the renosterveld soil samples than in cultivated fields. Notably, no significant difference in total nitrogen concentrations exists (Figure 6.15).

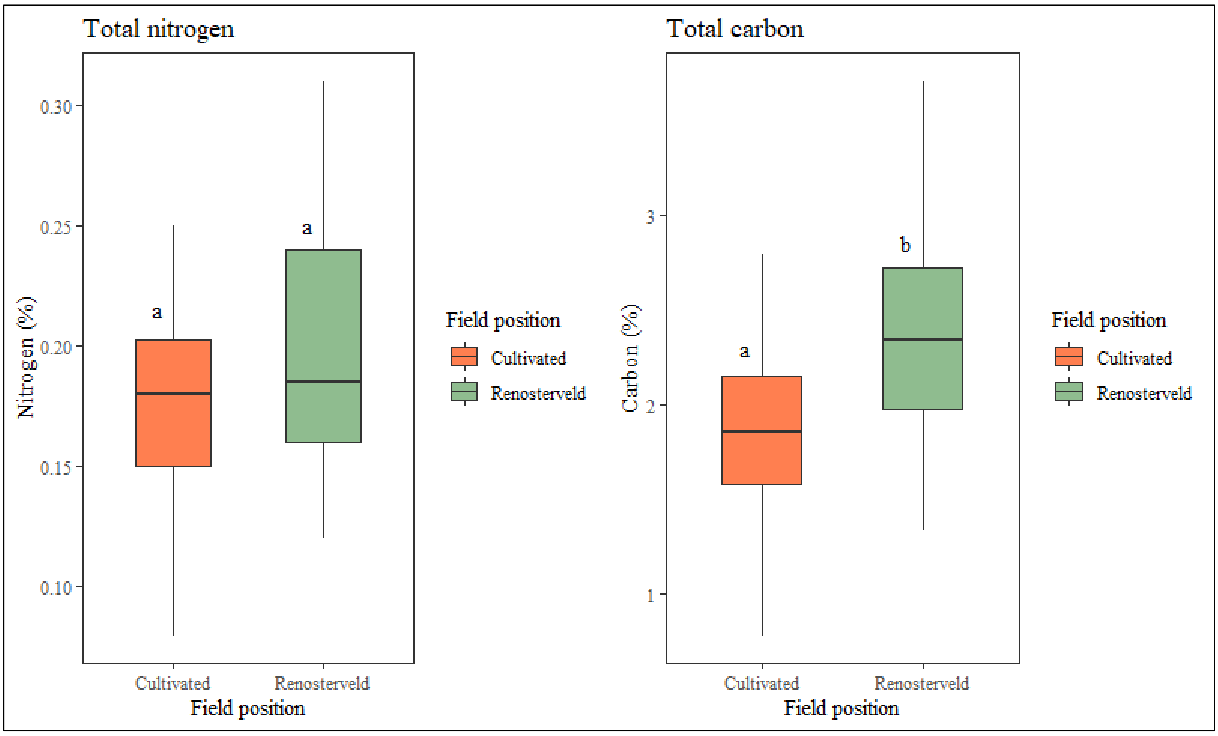


Figure 6.15 Boxplot comparing differences in total nitrogen and carbon in the soils collected from agricultural fields and surrounding natural vegetation. Median values with the same letters were not significantly different at $P \leq 0.05$.

While total N is not significantly different between cultivated fields and renosterveld soils, total C is. The question of whether these concentrations are derived from artificial fertilisation of soils or natural sources of N and C can be answered using isotopic analysis as discussed in Section 6.4.6.

6.4.6 Isotopic analysis of N and C in cultivated fields vs renosterveld soils

The isotopic composition of nitrogen and carbon in soils aids the determination of the sources of these elements in the soil. When considering the prevalence of $\delta^{15}N$ in each soil sample it is evident that renosterveld samples (shown in green in Figure 6.16) have lower levels of ^{15}N than cultivated field samples, while the concentration of total nitrogen in the soils appears not to affect the isotopic composition of nitrogen. Despite Transect 1 lying fallow for the year, cultivated field samples still have a lower $\delta^{15}N$ than those of renosterveld soils. However, the difference between these soils appears to be less than in Transects 2 to 4.

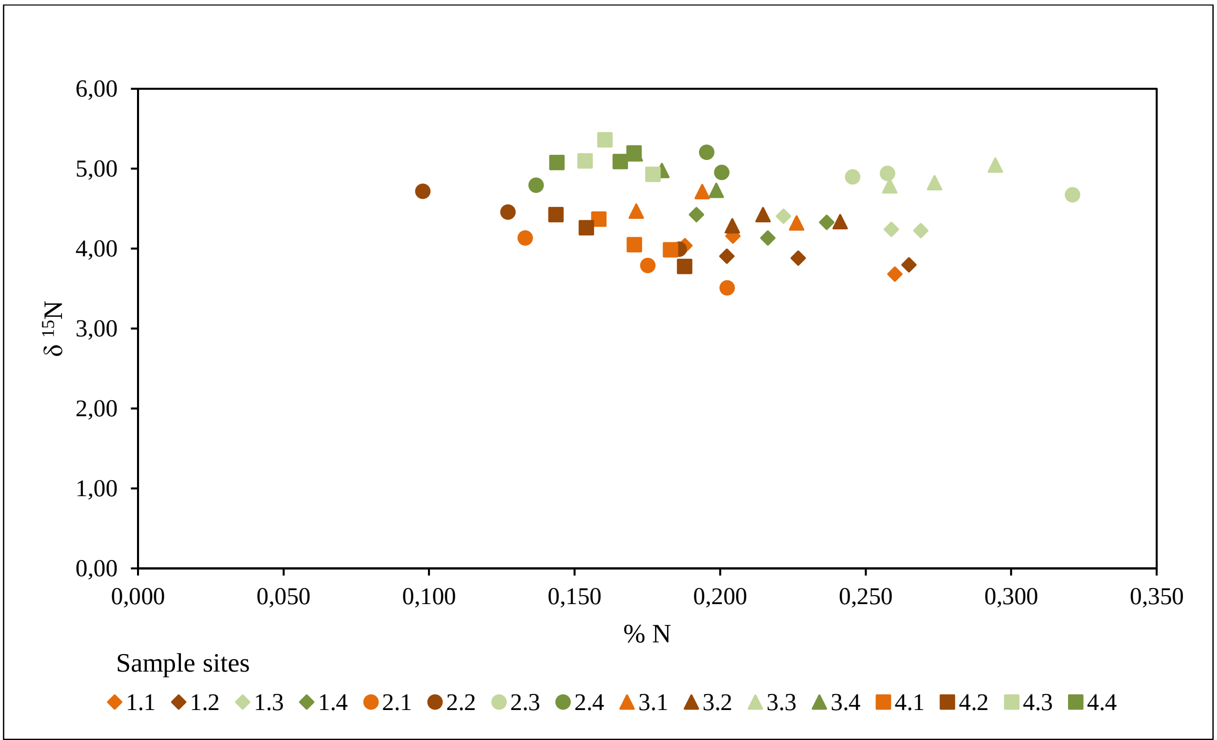


Figure 6.16 Influence of nitrogen concentrations in the soil on the isotopic composition of nitrogen for cultivated field (orange) and renosterveld (green) samples.

Temporally, no pattern exists concerning a changing isotopic composition on nitrogen in soils in response to fertilisation (Table 6.5). This indicates that the fractionation process of nitrogen is not altered by leaching downslope and that the addition of fertiliser results in a bulk lowering of ^{15}N:^{14}N in agricultural fields but no systematic behaviour is evident.

Table 6.5 $\delta^{15}N$ change over time in soil samples and triplicate samples of fertiliser applied to cultivated fields.

Site	**δ15N**		
	14 May	**28 May**	**28 June**
1.1	4.04	3.68	4.16
1.2	3.90	3.88	3.80
1.3	4.41	4.24	4.23
1.4	4.42	4.13	4.33
2.1	4.13	3.79	3.51
2.2	4.46	4.72	3.99
2.3	4.90	4.67	4.94
2.4	4.79	4.95	5.20
3.1	4.47	4.71	4.32
3.2	4.28	4.43	4.34
3.3	4.78	5.04	4.83
3.4	4.98	5.18	4.73
4.1	4.37	3.99	4.05
4.2	3.78	4.42	4.26
4.3	5.10	4.93	5.36
4.4	5.08	5.20	5.09
Fertiliser A	-1.55		
Fertiliser B	-1.53		
Fertiliser C	-1.50		

A statistically significant difference exists in the isotopic composition of nitrogen between cultivated fields and the adjacent renosterveld (P $\leq$0.05). However, for $\delta^{13}C$ (Figure 6.17), this pattern is not evident.

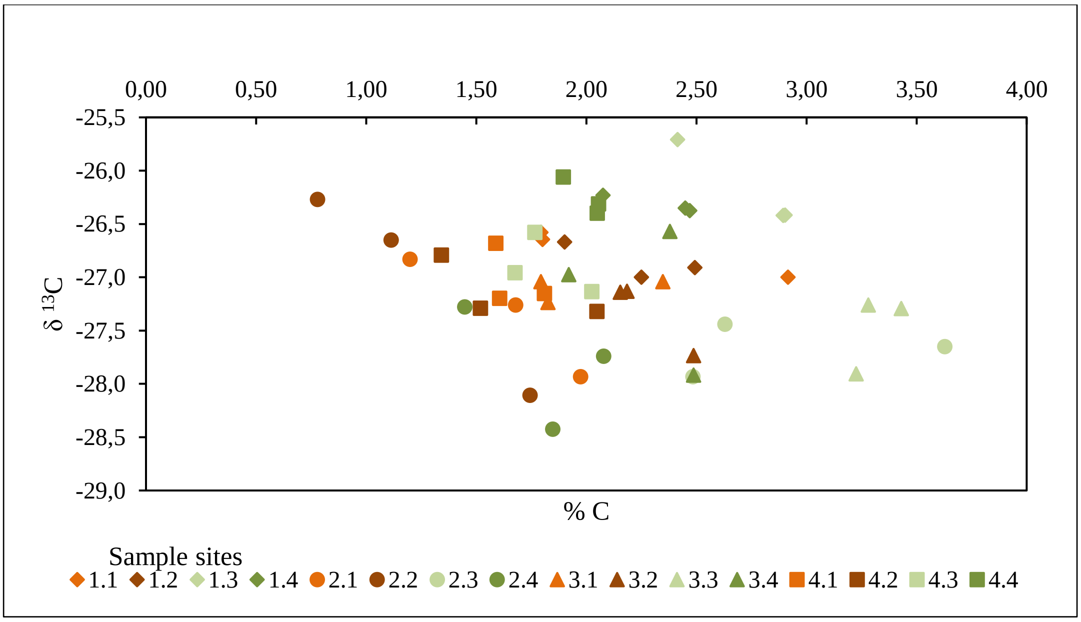


Figure 6.17 Influence of carbon concentrations of the isotopic composition of carbon for cultivated field (orange) and renosterveld (green) samples.

While the pattern of higher carbon concentration in renosterveld soils is evident (Figure 6.16) this has no significant influence on the overall isotopic composition of soil carbon itself principally because C is not a nutrient contained in the fertiliser applied to the cultivated fields (Table 6.1).

6.5 DISCUSSION

A discussion and interpretation of the results is presented below. First, the characterisation of the study sites' soils is considered (Section 6.5.1). Second, the spatio-temporal changes in nutrient concentrations between cultivated field and renosterveld soil samples is deliberated (Section 6.5.2), followed by an appraisal of the sources of these nutrient inputs into soils in the landscape (Section 6.5.3). Finally, a discussion on renosterveld contributions to ecosystem services in the landscape is presented (Section 6.5.4).

6.5.1 Soil characteristics

Soils in the study area vary in form, but have similar characteristics in terms of soil depth and texture. Soil EC_a differs significantly between cultivated field and renosterveld fragment conditions at both the 0.5-m and 1.0-m levels (recall Figures 6.4 to 6.6). Remaining natural vegetation patches often exist where soils are too shallow for cultivation, where rock banks are found below the surface or where slopes become too steep so limiting the ability to plough these areas (Donaldson et al. 2002; Topp & Loos 2019). The existence of subsurface rock banks is evident in all three transects (1.0-m depth) as indicated by the significant changes in EC_a (Figures 6.4 to 6.6), that partly explain the presence of adjacent renosterveld fragments. In terms of EC_a at 0.5-m depth, cultivated field and renosterveld soils are more homogenous.

Concerning soil texture, most A horizons for each transect are classified as clay loams with maximum depths ranging between 100 and 200 mm (Table 6.3), while B horizons are typically clay (Table 6.4). The B horizons for these sites have clay contents varying between 34.4% and 60.7%. These conditions are characteristic of texture-contrast (duplex) soils where the subsoil (B horizon) is finer than the A horizon (Hardie et al. 2012). Duplex soils are often cropped with wheat or lupins (Dracup, Belford & Gregory 1992) as presently done in the renosterveld landscape. The duplex soils here restrict root growth mechanically (as described by Dracup, Belford & Gregory 1992) by way of the bulk density of the soil in the B horizon. Duplex soils are plagued by a host of issues relating to cultivation and soil management, like soil saturation, crusting, restricted root growth (visible in both the cultivated and renosterveld fields in this study), desiccation, erosion by wind and surface runoff and high salinity (Hardie et al. 2012). Visual evidence gathered from the observation pits corroborates these claims where root depth is often limited by the B horizon in the soils.

In duplex soils, subsurface lateral flow (throughflow) results from infiltration of rainfalls through the A horizon that exceeds the hydraulic conductivity of the clayey B horizon (Hardie et al. 2012). Substantial amounts of phosphorus and nitrogen are able to move in subsurface lateral flow (throughflow) along the A-B horizon boundary in duplex soils (Hardie et al. 2012; Stevens, Cox & Chittleborough 1999) and nitrate losses can be ~21 times greater in throughflow than in overland flow (Cox & Ashley 2000; Cox & Pitman 2001). For each transect where a soil characterisation was performed the A horizon is bleached – a common feature indicating movement of water through this layer (throughflow). In Transect 2, it is expected that subsurface flow also takes place in the well-leached E horizon – a zone of eluviation[20]. Thus, the presence of these clayey strata (B horizons) clearly alters the vertical movement of water through soils and results in waterlogging and throughflow in the A and E horizons that lie above them (Anderson 1988). Furthermore, rapid saturation of the A horizon generates substantial overland flow, particularly on steeper slopes (Hardie et al. 2012).

6.5.2 Nutrient dynamics in soils

A statistically significant difference in soil phosphorus exists between the cultivated and renosterveld soils. Soil samples exhibit significantly higher plant-available phosphorus and total phosphorus concentrations in worked agricultural fields compared to natural vegetation fragments further downslope. Total phosphorus concentrations change in response to field fertilisation (Figure 6.9), even in the (unfertilised) natural vegetation remnants. This indicates that downslope movement of phosphorus does occur from the agricultural fields, especially after fertilisation events. This quite likely occurs through the sorption of phosphorus to fine colluvial materials such as clays (De Jonge et al. 2004).

P removal from agricultural fields and renosterveld fragments is evident by the difference in bioavailable P between cultivated fields and renosterveld fragments. Bioavailable P is typically soluble, whereas the remainder P (total P – bioavailable P) is largely immobile as it is insoluble and attached to organics or sediment (clays and silt) and not readily available for uptake by plants. Bioavailable P is being used by the renosterveld as shown in Figure 6.8, where renosterveld

[20] "The removal of soil material in suspension or solution from a part of or from the whole of a soil profile. The term leaching is preferred for removal in solution." (Van der Walt & Van Rooyen 1995, p.64)

fragments have a much lower mean and small variability in bioavailable P when compared to cultivated fields. The remainder of the P in these soils (~ 90% of total P) is attached to sediment and not available for use by plants. Therefore, the reduction in total P in renosterveld fragments is an indicator of two processes:

- A natural reduction in P as one moves downslope away from the source zone; and
- The effect of friction due to vegetation cover in the renosterveld fragments, which reduces flow velocity and limits the transport of sediment and organics with P bound to them.

In terms of ~90% of the total P measured in the system, the reduction in total P between cultivated fields and renosterveld fragments is not due to phytoremediation, but due to the buffers' effect of limiting downslope movement of sediments by increasing friction and reducing the runoff of sediment laden water. The reduction of bioavailable P in renosterveld buffers is encouraging as this indicates uptake of P by plants as they dissolved P moves downslope.

It is argued that the renosterveld adjacent to cultivated fields has significant hydrological and soil retention benefits, including increasing infiltration rates, reducing overland flow and resultant sediment loads, thus reducing loss of topsoil during rainfall events (O'Farrell, Donaldson & Hoffman 2009). Renosterveld buffers act to reduce topsoil loss by creating deposition zones as overland flow slows down, potentially limiting the transport of these nutrients to receiving waterbodies further downslope too. Phosphorus is often considered the most limiting nutrient in the Western Cape (Cramer 2010). Soil phosphorus concentrations in the fynbos biome range between 338 and 422 µg/g for total P and 13 to 40 µg/g plant-available phosphorus (Witkowski & Mitchell 1987) but soil P concentrations for these renosterveld soils ranged between 0 and 25 µg/g for plant-available phosphorus and 200 and 600 µg/g for total P. The results presented here indicate that these natural vegetation buffers are certainly receiving nutrient runoff from agricultural slopes as evidenced by the increase in total phosphorus concentrations post-fertilisation (Figure 6.9). Based on these results it is safe to conclude that renosterveld buffers assist in remediating these nutrients, as phosphorus levels in the natural veld are consistently lower than those of the agricultural fields, thus indicating uptake of bioavailable phosphorus by indigenous vegetation (Cramer 2010) and limiting the movement of insoluble P by reducing erosion of topsoil. These results corroborate the laboratory experiment performed by Jacklin, Brink & De Waal (2019; 2020).

Regarding nitrogen, there is, overall, no significant difference between cultivated field and renosterveld samples. There is an increase in total nitrogen concentrations after fertilisation in both the cultivated fields and renosterveld soils downslope (Figure 6.13). This suggests that downslope movement of nitrogen from the fertilised fields into renosterveld remnants (as evidenced by the increase in total P concentrations) does occur, likely as dissolved nitrogen in the soil and by overland flow (Billen, Garnier & Lassaletta 2013). Fertilisers (specifically nitrogen) might get used up quickly in these cultivated fields with crop growth or nitrogen is leached rapidly from cultivated fields at the onset of first rains in the region (Castellano, Lewis & Kaye 2013; Wilkison & Blevins 1999). Other sources of nitrogen, such as nitrogen fixation by bacteria, mineralisation of organic matter, animal waste and deposition from rainfall and the atmosphere in the natural vegetation might be present as well (Bloom 2015).

These results support the wealth of evidence given in the literature on the efficacy of natural buffers in remediating polluted waters (Cole, Stockan & Helliwell 2020). Buffer strips (and particularly grasses) are able to filter pollutants from overland flow by increasing surface roughness which, in turn, reduces flow rates of surface runoff and enhances infiltration into the soil substrate (Borin et al. 2005; Liu, Zhang & Zhang 2008). The resultant deposition of sediment and increased infiltration enhances the efficacy of pollutant and nutrient adsorption to soil and plant roots. Nutrients and pollutants are then taken up and phytoremediated by plants in this zone. Renosterveld buffers have significantly lower total P in their soils than the cultivated fields upslope, thereby demonstrating that these zones either absorb nutrient inputs from agriculture or there is little mobility of P between these hillslope sections. However, the increase in total P after fertilisation in renosterveld fields suggests that P is somewhat mobile at these sites by soil erosion and downslope movement of soluble (bioavailable P). Renosterveld buffers show great potential for reducing pollutant delivery rates from source to watercourses, similar to that of other vegetation types (Arora, Mickelson & Baker 2003; Arora et al. 1996; Liu, Zhang & Zhang 2008) by filtering out particulates and dissolved nutrients (Stutter, Chardon & Kronvang 2012; Stutter, Langan & Lumsdon 2009; Tanner, Nguyen & Sukias 2005). However, most of these studies are based on vegetation buffers in the riparian zone (Stutter, Chardon & Kronvang 2012), whereas research on phytoremediation by renosterveld species focuses on common wetland plants (Jacklin, Brink & De Waal 2019; 2020). The results presented here were not based on riparian vegetation, rather on any renosterveld fragment located downslope of cultivated fields and they demonstrate the value of these renosterveld remnants, even if not located in the riparian zone. This is vitally important as riparian zones in the renosterveld landscape are often dominated by invasive species or they have been converted to grazing for livestock and have thus been degraded, while the more intact fragments further upslope can still perform this valuable function.

The results also show that total carbon is significantly higher in renosterveld fragments than in cultivated fields (Figure 6.15). Soil carbon levels are typically affected by plant biomass (Wan et al. 2019) and they markedly increase soil quality by improving water and nutrient retention and decreasing erosion (Ontl & Schulte 2012). Renosterveld fragments are characterised by low shrubs and grasses. In contrast, the agricultural fields are ploughed, planted, harvested and left fallow (bare, unplanted) annually, leaving limited biomass to enrich the soils. Remaining post-harvest biomass in the fields is further grazed by sheep. The greater carbon concentration in renosterveld fragments (Figure 6.15) signifies that they store significantly more carbon than agricultural fields, so corroborating the findings reported by Mills et al. (2013) and offering another valuable ecosystem service – carbon sequestration.

6.5.3 Nitrogen sources

The $\delta^{15}N$ concentrations in the samples from cultivated fields are lower than renosterveld samples. As fertilisers are artificially (man-made) produced, they typically have lower $\delta^{15}N$ values. The samples analysed in this study had a $\delta^{15}N$ of ~ -1.53. The $\delta^{15}N$ in natural soils is typically higher as shown in the results presented here. The bulk application of fertilisers to agricultural fields has a profound influence clearly evidenced by cultivated field $\delta^{15}N$ values being significantly lower than those of the renosterveld samples. A major source of nitrogen in these cultivated fields is undoubtedly fertiliser application. No temporal pattern exists concerning a changing isotopic composition on nitrogen in soils in response to fertilisation, an indication that the fractionation process of nitrogen is not altered by leaching downslope and that the addition of fertiliser results in a bulk lowering of $\delta^{15}N$ in agricultural fields (Choi et al. 2006), but no systematic behaviour is evident.

6.5.4 Contribution to ecosystem services in the renosterveld

The renosterveld landscape is notable for the loss of biodiversity and fragmentation of vegetation units (Curtis, Stirton & Muasya 2013; Donaldson et al. 2002; Kemper, Cowling & Richardson 1999). The loss of natural vegetation over time has limited the provision of ecosystem services in the landscape. This study points to the value of renosterveld fragments in carbon sequestration and corroborate those of Mills et al. (2013) who found that active (farmed) fields (69 Mg C/ha) store less carbon than renosterveld fragments (84 Mg C/ha) (Mills et al. 2013). The results of this research also strongly indicate that renosterveld buffers significantly reduce soil phosphorus (Figures 6.7 and 6.8) applied to cultivated fields as phosphorus moves downslope into renosterveld zones. Nitrogen may leach too readily and movement downslope may be too quick (low residence time) for phytoremediatory processes in the renosterveld to be effective. In this case, greater buffer width and slopes of lower gradients may help with the remediation of nitrogen in the renosterveld environment. Global literature reveals the potential of vegetation buffers to remediate pollution from agriculture (Borin et al. 2005; Lee et al. 2000), but many of these studies test only the potential of riparian vegetation to do so (Allaire et al. 2015; Hoffmann et al. 2009; Stutter, Chardon & Kronvang 2012). These results show that, in renosterveld, even non-riparian fragments are able to remediate phosphorus loading from agricultural slopes and the effect of buffer width and hillslope gradient on remediation of nitrogen is an avenue for further research.

6.6 CONCLUSIONS

This study demonstrates that the existence of indigenous renosterveld buffers between agricultural fields and waterbodies significantly reduce P mobility, thus offering a valuable ecosystem service in terms of improving water and soil quality. The results show that soils respond to fertilisation, with total phosphorus levels increasing markedly post-fertilisation in fields and in natural vegetation patches downslope. Thus, these fragments receive nutrients from upslope areas. However, agricultural fields contain significantly higher levels of phosphorus than natural vegetation fragments further downslope indicating uptake of these nutrients by the natural vegetation (which is often P limited). No significant difference in total N levels between renosterveld soils and cultivated ones exists. While total P and N levels increased in renosterveld soils post-fertilisation – indicating nutrient movement downslope – renosterveld soils still have significantly lower concentrations of P specifically indicating abstraction of nutrients by these buffers or that P was not mobilised from cultivated fields into these fragments.

The application of NPK fertilisers on agricultural slopes affects the isotopic composition of nitrogen in agricultural fields when compared to surrounding natural vegetation. $\delta^{15}N$ is significantly more prevalent in natural soils than in those actively fertilised, indicating a bulk lowering of ^{15}N:^{14}N as a result of fertiliser application. Soil carbon is significantly higher in renosterveld soils than in cultivated fields, which results in a serious impact on the carbon budget of the renosterveld landscape. Landuse change in the renosterveld has thus also contributed to climate change by removing a valuable carbon store, as seen in many landscapes worldwide. Buffer strips can positively protect the local environment and waterbodies from nutrient loading as seen globally. What is most encouraging is the apparent ability of non-riparian vegetation to remediate this pollution in the renosterveld.

The conservation of renosterveld fragments is of crucial importance, not only for the biodiversity of this endangered vegetation type but because it also offers great potential for ecosystem services in terms of erosion reduction, limiting soil transport (Chapter 4) and remediating nutrient loading from agriculture as shown by the reduced phosphorus concentrations in the fragments, though the impact of renosterveld buffers on nitrogen requires further investigation.

CHAPTER 7: SYNTHESIS AND CONCLUSION

7.1 INTRODUCTION

This study was designed to investigate the impacts of agricultural transformation on the hydrology, sediment and nutrient dynamics in a renosterveld landscape, with a focus on the Bot River catchment in the Overberg region of the south-western Cape. The purpose of the research was to fill identified research gaps by demonstrating how landuse change has affected catchment hydrology and soil erosion in the renosterveld landscape as well as the degree of nutrient loading in the Bot River. A further objective was to determine the potential of renosterveld buffers to remediate pollution from agricultural hillslopes.

The study aimed to add to the body of knowledge relating to ecosystem services offered by renosterveld vegetation. Renosterveld is a well-studied field with species diversity, rehabilitation and habitat as well as ecological functioning as three central themes of investigation. Only a few studies have investigated the biogeochemical–landscape–hydrological relationships or ecological services provided by the biome (e.g. Egoh et al. 2008; Reyers et al. 2009) so that a paucity of literature exists, particularly regarding an understanding of the impact of natural vegetation on hydrological functioning and nutrient dynamics. Extant research on the ecosystem services offered by renosterveld focuses on the salinisation of local soils, the potential for carbon sequestration by renosterveld fragments and lower infiltration of rainfall into soils in transformed fields (O'Farrell, Donaldson & Hoffman 2009). Several research gaps were identified that informed the objectives of this study. First the study evaluated how landuse change has affected catchment hydrology across the renosterveld landscape (Objective 1). Second, the impact of cultivated fields on soil erosion and sediment delivery to river systems in the Overberg was investigated (Objective 2). Third, the spatio-temporal impacts of fertiliser application to cultivated fields on the freshwater systems in the Overberg were quantified (Objective 3) and finally, the viability of renosterveld buffers for remediating water quality issues in the Overberg landscape was determined (Objective 4).

In this chapter the study's findings are summarised, conclusions are drawn based on the findings of the modelling and empirical studies conducted in Chapters 3-6. First the impacts of agriculture on the hydrology and sediment loads in the Overberg are discussed (Section 7.2). Second, the nutrient levels in the Bot River and their relationship to agricultural landuses are considered (Section 7.3). Third, the phytoremediation ecosystem service delivery by renosterveld fragments is summarised and fourth, the contributions of the research to conservation objectives in the renosterveld are considered (Section 7.5). Finally, the study's novelty (Section 7.6) as well as study limitations and further research avenues are discussed (Section 7.7).

7.2 IMPACTS OF AGRICULTURE ON THE HYDROLOGICAL FUNCTIONING AND SEDIMENT DYNAMICS IN THE OVERBERG (OBJECTIVES 1 AND 2)

The landscape in the Overberg region has undergone substantial agricultural transformation from natural vegetation to cultivated fields. The impacts of this landuse change have fallen hardest on Rûens Shale Renosterveld, with only 4-10% of its original spatial extent remaining (Rebelo et al. 2006). Hydrological processes (both surface and groundwater), sediment delivery and pollution of waterbodies have been altered by these changes in landuse.

The results of the study show that changes in landuse greatly affect the hydrological functioning of catchments at a landscape scale in the Overberg (Objective 1) by altering the physiographic similarity of small catchments in the region. The quinary catchments delineated and then clustered together based on the physiographic characteristics of each catchment showed marked change to cluster membership based on the degree of landuse change experienced by each catchment. It was anticipated that catchments within the same cluster would behave similarly to one another and exhibit a similar runoff response to rainfall. The landscape level analysis showed that landuse change greatly affects the physiographic similarity of quinary catchments as their cluster membership changes when the landuse input is altered from natural vegetation to contemporary landuse designations. The physical characteristics of each catchment (e.g. elevation range, soil-clay content, slope, catchment area and soil depth) are shown to be major determinants of cluster membership (physiographic similarity), but landscape transformation is similarly important. Agricultural transformation of the Overberg was found to demonstrably affect hydrological dynamics on a landscape level. Thus, a study area (Bot River) was selected to determine the impact of landuse change on a catchment scale (Objective 2 – Chapter 4).

The Bot River catchment is relatively data-poor, with only one streamflow gauge and limited meteorological data. However, a fully distributed JAMS/J2000 streamflow model was generated for the catchment using a combination of remotely sensed data inputs (digital surface model and landuse data sets) coupled with observed and modelled meteorological inputs. The model was used to approximate the relative flow components of and simulate sediment delivery for the Bot River under three landuse and two climatic scenarios. This enabled a quantification of the impact of landuse and climatic variability on streamflows and soil erosion in the Bot River (Objective 2). The results recorded a C_v value of 0.54 for the Bot River, which is indicative of moderate to high inter-annual flow variability in the catchment, particularly for a perennial river. The flow variability is, however, below that of many southern African catchments which typically have a C_v value of 0.81 (Finlayson & McMahon 1988). Direct surface runoff was found to be the greatest contributor to streamflow in the wet, winter months. However, during November to April, streamflow is maintained by baseflow contributions from primary and secondary aquifers. The clay-enriched subsoil (B horizon) is known to be a limiting factor to water percolation through the soil and results in relatively larger overland flow components when compared to infiltration and subsurface flows (see Section 6.5.1), which might explain these results. It results also in high throughflow as mentioned in Chapter 6.

The results demonstrate the impact of landuse change on the hydrology and sediment delivery of the catchment where modelled mean annual sediment yield for the catchment is reported at 423 t/km^2/yr for 1990 increasing to 490 t/km^2/yr for 2018 landuse. These results are comparable with regional data of Msadala et al. (2010), Msadala & Basson (2017), Rooseboom (1975), and Rooseboom et al. (1992) who all report similar regional values. The results presented in Chapter 4 also concur with evidence presented by Msadala et al. (2010) and Msadala & Basson (2017) suggesting that soil erosion in South Africa is increasing. The modelling showed that soil erosion in the catchment is much more prevalent in low-lying foothills, characterised by agricultural landuses, limited soil conservation practices and readily erodible soils than in areas of high elevation in the catchment. It is evident that soil erosion is increasing in the catchment due to landuse change. Furthermore, the results for 1990 and 2018 landuses are markedly higher than simulated sediment yield under natural vegetation conditions (22 t/km^2/yr).

Increased sediment loads deliver nutrients, particularly phosphorus, and pollutants to waterbodies (Kronvang, Laubel & Grant 1997; Schoumans et al. 2014). Therefore, landuse changes often affect nutrient dynamics in rivers, which was explored in Chapter 5 (Objective 3).

7.3 NUTRIENT LEVELS IN THE BOT RIVER DUE TO AGRICULTURAL LANDUSE (OBJECTIVE 3)

Agriculture is one of the leading causes of water degradation globally through the incidental release of agrochemicals and nutrients (Sharpley et al. 2015; Wang et al. 2018; Nie et al. 2018) from adjacent hillslopes into receiving bodies (Evans et al. 2019). The bulk of nitrogen and phosphorus transfer in agricultural regions arises from the mobilisation of nitrogen and phosphorus-based fertilisers and manure. Agriculture was found to affect water quality by exacerbating sediment inputs into rivers (Objective 3 in Chapter 4), while also reducing flows due to water extraction for crop irrigation. The Bot River was found to follow these trends, with nutrient loading associated with agricultural landuses in the catchment.

The Bot River system was found to display high EC and Cl^- concentrations across most of the sampled sites with the exception of mountain streams, which had substantially lower EC and Cl^- levels than foothill and lowland river reaches. EC and Cl^- concentrations were found to be affected by the input of water from mountain streams (underlain by sandstone geology) which reduce EC and Cl^-. Shale tributaries were found to increase the levels substantially because the loose materials above underlying bedrock (Rûens shale) in the area record high levels of stored salts that have accrued over time. It was concluded that inputs from foothill and lowland areas contribute greatly to overall salinity in the Bot River system.

Nitrogen levels were found to vary spatially and temporally in the river. The mountain streams consistently displayed lower nitrate levels than lowland areas because the mountain streams are adjacent to natural vegetation, whereas the river reaches, surrounded by cultivated, dryland agriculture, have substantially higher nitrogen levels. This demonstrates the link between agriculture and nitrogen loading of river systems. This relationship was further illustrated by the seasonality of nutrient fluxes in the system. Nitrogen levels increased by an order of magnitude or greater in response to winter rainfalls when compared to the dry summer months. This was found to be directly related to the fertilisation of agricultural fields in the area during May and is seen in the spatial distribution of nutrient loads in the river.

Regarding phosphorus, SRP was not shown to be an effective measure of phosphorus levels in the river as the results were often below detectable limits. However, the Swart River was found to be a major point source of phosphorus input into the Bot River. As this tributary drains an area dominated by rain-fed agriculture (primarily wheat and oilseed crops) the relationship between landuse transformation and nutrient dynamics is again evident. In response to landuse change and agricultural practices the Bot River's trophic level appears to have increased when compared to conditions in 1985 (Bally & McQuaid 1985) and 2004/2005 (Herdien et al. 2006).

7.4 ECOSYSTEM SERVICES – DOES RENOSTERVELD OFFER A SOLUTION? (OBJECTIVE 4)

As landuse transformation has demonstrably affected streamflow, sediment and nutrient dynamics in the Bot River area, attempts should be made to mitigate these impacts. A solution is the use of natural vegetation buffers to remediate polluted waters from hillslopes before they enter rivers (Objective 4). Laboratory-based research suggested that wetland plants based in the renosterveld landscape can effectively remove nitrogen and phosphorus from water and sediment through phytoremediation (Jacklin, Brink & De Waal 2019; 2020). This research successfully demonstrated that renosterveld fragments have lower P and higher C concentrations than cultivated fields further upslope (Objective 4).

Soil samples were found to exhibit significantly higher plant-available phosphorus and total phosphorus concentrations in cultivated fields when compared to those in natural vegetation fragments further downslope. Total phosphorus concentrations were observed to change in response to field fertilisation, even in the (unfertilised) natural vegetation remnants, so demonstrating that downslope movement of phosphorus does occur from the agricultural fields (especially after fertilisation events). It very likely occurs through the sorption of phosphorus to fine colluvial materials such as clays. Phosphorus levels were found to drop significantly within the adjacent renosterveld soils, an indication of either limited movement of phosphorus from agricultural fields or substantial uptake of phosphorus from renosterveld vegetation.

Renosterveld buffers are known to reduce topsoil loss by creating deposition zones as overland flow slows down – potentially limiting the transport of these nutrients to receiving waterbodies further downslope as well. Phosphorus is often considered the most limiting nutrient in the Western Cape, so it is not surprising that this is reduced in the natural vegetation where it is rapidly taken up when made available. Regarding nitrogen, no significant difference was found between cultivated field and renosterveld samples but there does appear to be an increase in total nitrogen concentrations after fertilisation in both the worked agricultural fields and natural vegetation downslope. This suggests that there is downslope movement of nitrogen from the fertilised fields into renosterveld remnants, most likely as dissolved nitrogen in the soil (throughflow) and by overland flow. It is concluded that fertilisers (nitrogen specifically) might get used up quickly, both in cultivated fields and in natural vegetation, or they can be transported rapidly downslope at the onset of first rains in the region. Other sources of nitrogen, such as nitrogen fixation by bacteria, mineralisation of organic matter, animal waste and deposition from rainfall and the atmosphere in the natural vegetation may be present as well.

Soil organic matter is known to substantially increase soil quality by improving water and nutrient retention and by decreasing erosion. It was found that total carbon is significantly higher in renosterveld fragments than in cultivated fields. Soil carbon levels are typically affected by plant biomass and the remnants of natural renosterveld vegetation are characterised by low shrubs and grasses. In contrast, the agricultural fields are ploughed, planted, harvested and left fallow (bare, unplanted) annually, leaving limited biomass to enrich the soils. The remaining post-harvest biomass in the fields is further grazed by sheep. The higher carbon concentrations in renosterveld fragments indicate that they offer significant ecosystem services. As soil carbon has a strong, positive correlation with increased infiltration rates and lag time before runoff (after rainfall starts)

(O'Farrell, Donaldson & Hoffman 2009), the renosterveld fragments serve to decrease erosion and create sediment accumulation zones beneath agricultural hillslopes. These findings support the wealth of evidence in the literature about the efficacy of natural buffers in remediating polluted waters.

7.5 INFORMING THE CONSERVATION OBJECTIVES FOR THE OVERBERG RENOSTERVELD

The remaining renosterveld in the Overberg is often found in areas adjacent to watercourses and drainage lines that have not been ploughed. These fragments are valuable ecological corridors for animal and plant species. Animal activity in the corridors is high, with small- to medium-sized antelope, small predators and many endemic bird species all present. Without these renosterveld fragments the natural Overberg landscape would be largely lifeless. The fragments have therefore been identified as crucial objects for conservation in the renosterveld landscape. The Overberg Renosterveld Conservation Trust (ORCT) has established a watercourse restoration project to prioritise the conservation of these corridors (Curtis 2016).

Four priorities have been identified for renosterveld conservation: governance and formal protection, agricultural practices and incentives, the management of renosterveld fragments and informing and managing perceptions (Topp & Loos 2019). Formal protection of renosterveld vegetation envisages the purchase of land covered by renosterveld vegetation for conservation. Management of agricultural activities and practices involves mitigating overgrazing and the development of management plans to prioritise the expansion of farming activities into fallow fields and former agricultural zones instead of into areas of high species diversity. The management of renosterveld fragments should concentrate on species heterogeneity that allows for improved pollinator numbers as well as enlarging fragment sizes and identifying priority clusters such as river corridors (Von Hase et al. 2003).

The greatest challenge in terms of conservation is the management of perceptions around the value of renosterveld. Farmers' and landowners' attitudes to renosterveld conservation are often negative or apathetic as the vegetation is not perceived to be economically beneficial (Winter, Prozesky & Esler 2007). Unfortunately, many of the remaining renosterveld fragments are located on private property, which inhibits conservation efforts. The ORCT has actively engaged with local farmers in the Overberg through the conservation easement programme that allows farmers to insert a restriction in their title deeds demarcating a certain portion of their land as a conservation servitude in perpetuity (Curtis-Scott 2020). Improved participation in this programme can certainly contribute to the conservation of remaining renosterveld.

The provision of information about the beneficial ecosystem services delivered by renosterveld fragments can give strong support to conservation efforts by informing farmers of the hidden value of the fragments.

The findings of this research show how renosterveld fragments contribute to ecosystem services in the landscape by reducing erosion (Chapter 4) and purifying water (Chapter 6) before it reaches watercourses. These revelations should make valuable contributions to changing landowners' perceptions of the value of renosterveld vegetation. The study has produced incontrovertible evidence that renosterveld corridors do impact positively on water quality in the landscape through the remediation of fertilisers that move downslope before entering watercourses. It is imperative that these corridors be conserved and restored to improve the serviceable effect of these buffer strips.

7.6 STUDY NOVELTY

This study contributes novel research located in a critically endangered vegetation landscape, renosterveld. The novelty exists in three areas. First, the research presents an innovative approach to simulating streamflow and sediment dynamics in data-sparse areas by using a fully distributed J2000 rainfall–runoff model. This is the first fully distributed rainfall–runoff model developed for the region and the first results that confirm and quantify the impacts landuse change have on soil erosion in the Overberg. Remotely sensed data (landuse and a digital elevation model (DEM)) are combined with only a few measured data sets (streamflow and meteorological inputs) within the J2000 environment to produce different scenarios of sediment delivery. This is equally applicable elsewhere (especially in other data-poor areas) using the JAMS/J2000 environment and the testing in the Bot River catchment demonstrates how soil erosion is impacted by landuse and climatic variability. The approach can also be used for future analyses of climate change projections and the effects of increased rainfall intensity on soil erosion in this and other catchments.

Second, new, high-resolution spatial data on the water quality of the Bot River were collected where very little existed previously. These data are used to assess the changes in nutrient status of the river over time (in response to agricultural inputs) and to illustrate how nutrient loads in the catchment are related to landuse. These data also confirm that nutrient dynamics in a river are site (reach) specific and that the appropriate monitoring of systems requires more than just one observation site within a catchment. The results also speak to the optimal location of appropriate monitoring sites along a river's long profile. Priority must be given to sites downstream of potential point sources (such as low-quality tributaries) and river reaches with high nutrient loads. The study also sheds new light on the impact of agricultural activities on water quality in a semi-arid, shale-dominated landscape and adds original findings to the literature on drylands agriculture. A noteworthy contribution is the finding that under-monitoring in South African rivers limits the quality and completeness of the data provided to international data repositories and impacts our understanding of the global status of water quality.

Third, this study is the first field-based evaluation of the impact of renosterveld vegetation buffers on nutrient dynamics in the Overberg region. This positive should be used to encourage the preservation of existing renosterveld fragments as these fragments deliver valuable ecosystem services and they help to conserve water quality in these regions. Although some studies have assessed the impacts of renosterveld vegetation on infiltration rates and soil retention (e.g. O'Farrell, Donaldson & Hoffman 2009) and on carbon sequestration (e.g. Mills et al. 2013), none have determined the potential for this type of vegetation to remediate nutrient loading from agricultural hillslopes. The methods used in this study can be used in other contexts and areas as they are applicable to other river systems and landscapes in South Africa, and internationally too.

7.7 LIMITATIONS AND FURTHER RESEARCH

A limitation to this study was its restriction to a case study of the Bot River in the wider Overberg region. The spatial extent of the region is too great for undertaking detailed analyses of each river system which would have been the preferable approach. While the finding related to Objective 1 indicate that landuse changes affect the hydrology of the entire Overberg, the transferability of the findings made in researching Objectives 2, 3 and 4 to other catchments in the region needs to be explored. Furthermore, time, travel and monetary constraints limited the type and volume of data (samples) that could be collected and analysed for the Bot River (Objective 3) and soils in the catchment (Objective 4). Water samples were only collected *after* large storm events so further research should be conducted on the nutrient response of the river to a specific storm in order to determine the lag time between rainfall onset and nutrient response. Similarly, a higher temporal resolution to evaluate how rapidly soil chemistry changes in response to fertilisation warrants further investigation. Since this study was spatially confined, the methods developed and used should be applied to other areas in the Overberg renosterveld to determine whether there is a consistent pattern to soil erosion and nutrient loads, as well as the efficacy of different renosterveld assemblages on phytoremediation.

7.8 CONCLUSION

The findings demonstrate the impact of agricultural expansion on the hydrology, sediment and nutrient dynamics of the Overberg region in the Western Cape. Shifts in landuse from indigenous renosterveld vegetation to cultivated oilseed and cereal crops have resulted in intensified soil erosion and increases in nutrient delivery to watercourses from fertilisers applied to cultivated hillslopes. To mitigate these impacts the conservation and restoration of renosterveld buffers should be prioritised. Not only do these buffers offer a valuable refuge for indigenous fauna and are the home to striking floral diversity, they also offer a valuable ecosystem service by carbon sequestration and limiting the movement of P downslope. Their efficacy in terms of remediating nitrogen pollution warrants further investigation, with the specific impact of buffer width and slope being key factors for consideration. The findings of this research add to a deeper understanding of the hydrology and sediment dynamics of small, dryland river systems as well as to the adverse influence landuse conversion to cultivated fields has on nutrient pollution of rivers. It is especially noteworthy that this study is the first to show that remediation of phosphorus from hillslopes is feasible where renosterveld fragments are left, thus offering viable solutions to some of the existential challenges facing this dryland landscape.

[Total word count: 78450]

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APPENDICES

APPENDIX A: LANDUSE RECLASSIFICATION

Table A1 Reclassified 2018 landuse classes with their 1990 analogies. Classification performed based on hydrological response of each class.

Landuse classes (1990)	Reclassified landuse classes (2018)
Plantations and Fynbos (thicket)	Contiguous dense plantation forest
	Contiguous low forest (thicket)
	Dense forest (woodlands)
	Fallow old fields (trees)
	Smallholdings (tree)
	Temporary unplanted forests (felled)
Cultivated annual (non-pivot)	Commercial annual crops (non-pivot irrigated)
	Commercial annual crops (dryland)
	Subsistence/small-scale annual crops
Cultivated annual - pivot	Commercial annual crops (pivot irrigated)
Cultivated permanent orchards	Cultivated commercial permanent orchards
Cultivated permanent vines	Cultivated commercial permanent vines
Settlements	Commercial
	Industrial
	Residential formal (bare)
	Residential formal (low vegetation/grass)
	Residential formal (tree)
	Residential informal (bare)
	Residential informal (low vegetation/grass)
	Residential informal (tree)
	Roads, rails (major linear)
	Urban (bare)
	Urban (grass)
	Urban (tree)
	Village (dense)
	Village (scattered)
Wetlands	Fallow old fields (wetlands)
	Herbaceous wetlands (currently mapped)
	Herbaceous wetlands (previously mapped)
Fynbos (open bush and low shrubland)	Fallow old fields (low shrubland)
	Fynbos (low shrubland)
Fynbos (grassland)	Fallow old fields (grass)
	Natural grasslands
	Smallholdings (low vegetation/grass)
Fynbos (bare ground)	Bare riverbed material
	Coastal sand dunes
	Dry pans
	Fallow old fields (bare)
	Natural rock surfaces
	Other bare
	Smallholdings (bare)

Mines	Mines (extraction pits, quarries)
Waterbodies	Artificial dams
	Artificial flooded mine pits
	Artificial sewage ponds
	Natural estuaries
	Natural ocean (coastal)
	Natural pans (flooded at observation)
Degraded	Eroded lands

APPENDIX B: BOT RIVER ION CONCENTRATIONS AND ISOTOPE DATA

Table B1 Anion and cation concentrations for each sample site over the year-long monitoring of the Bot River.

Date	F^- (mg/l)	Cl^- (mg/l)	SO_4^{2-} (mg/l)	PO_4^{3-} (mg/l)	NO_2^- (mg/l)	Br^- (mg/l)	NO_3^- (mg/l)	NO_3-N (mg/l)	NO_2-N (mg/l)	NO_x-N (µg/l)	Li^+ (mg/l)	Na^+ (mg/l)	NH_4^+ (mg/l)	K^+ (mg/l)	Mg^{2+} (mg/l)	Ca^{2+} (mg/l)
Site 1																
2020/10/15	0.177	301.489	56.382	0	0	1.063	2.385	0.54	0.00	538.76	0	146.724	0	3.412	33.755	28.407
2020/11/26	0.221	399.965	35.856	0	0	1.428	0	0.00	0.00	0.00	0	195.124	0	3.175	46.377	35.413
2020/12/15	0.267	498.775	14.471	0	0	1.786	0	0.00	0.00	0.00	0.024	237.749	0	3.324	54.432	38.951
2021/01/15	0.346	508.736	10.207	0	0	1.876	0.226	0.05	0.00	51.05	0.04	245.719	0	2.54	56.495	40.324
2021/02/17	0.182	291.721	29.842	0	0	1.019	0	0.00	0.00	0.00	0	143.802	0	2.962	36.462	33.029
2021/03/16	0	250.746	25.429	0	0	0.923	0	0.00	0.00	0.00	0	120.66	0	3.562	31.613	29.576
2021/04/15	0.17	239.108	29.121	0	0	0.881	0	0.00	0.00	0.00	0	117.815	0	2.155	31.446	28.526
2021/05/11	0.187	449.872	56.904	0	0	1.45	6.256	1.41	0.00	1413.21	0	214.704	0	5.262	53.701	51.359
2021/05/26	0.165	531.584	80.611	0	0	1.656	9.868	2.23	0.00	2229.15	0	255.466	0	5.16	64.719	60.342
2021/06/28	0.194	340.928	45.269	0	0	1.023	4.442	1.00	0.00	1003.43	0	165.354	0	4.414	38.663	35.309
2021/07/07	0.127	239.659	52.656	0	0	0.755	9.096	2.05	0.00	2054.76	0	120.081	0	3.701	32.3	32.903
2021/08/02	0.124	223.695	47.02	0	0	0.702	5.809	1.31	0.00	1312.23	0	103.146	0	3.107	25.51	24.806
2021/08/31	0.172	200.067	46.444	0	0	0.624	5.517	1.25	0.00	1246.27	0	92.202	0	2.876	23.522	21.696
2021/09/22	0.117	233.404	43.537	0	0	0.748	3.239	0.73	0.00	731.68	0	107.339	0	2.269	27.123	23.554
Site 2																
2020/10/15	0.23	246.683	48.849	0	0	0.723	5.473	1.24	0.00	1236.33	0	131.701	0	2.141	28.433	24.72
2020/11/26	Not available (NA)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2020/12/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/01/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/02/17	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/03/16	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/04/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/05/11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/05/26	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

2021/06/28	0.153	201.56	19.272	0	0	0.416	0.535	0.12	0.00	120.85	0	87.01	0	1.796	16.328	12.582
2021/07/07	0.193	166.131	62.297	0.307	0.206	0.438	46.684	10.55	0.06	10608.48	0	93.215	0.035	4.561	25.499	34.524
2021/08/02	0.206	201.654	64.735	0	0.332	0.527	31.599	7.14	0.10	7239.19	0	102.981	0	3.16	26.653	31.16
2021/08/31	0.24	244.755	65.704	0	0	0.66	25.299	5.71	0.00	5714.96	0	126.263	0	3.202	32.334	35.524
2021/09/22	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Site 3																
2020/10/15	0.046	29.743	3.502	0	0	0.066	0.31	0.07	0.00	70.03	0	15.415	0	0.562	4.022	5.359
2020/11/26	0.063	88.74	18.426	0	0	0.335	0.499	0.11	0.00	112.72	0	45.705	0	1.37	8.506	6.988
2020/12/15	0.194	108.324	12.449	0	0	0.47	0.703	0.16	0.00	158.81	0	52.644	0	3.194	9.343	7.965
2021/01/15	0.11	89.987	6.988	0	0	0.408	0.38	0.09	0.00	85.84	0.017	44.94	0	1.758	7.327	6.403
2021/02/17	0.122	70.578	10.327	0	0	0.218	4.13	0.93	0.00	932.95	0	35.81	0	2.747	5.761	6.274
2021/03/16	0.098	95.052	6.917	0	0	0.421	1.367	0.31	0.00	308.80	0	47.097	0	2.298	7.883	6.478
2021/04/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/05/11	0.122	256.663	101.394	0	0	0.756	38.295	8.65	0.00	8650.71	0	147.281	0.097	4.065	33.725	33.164
2021/05/26	0.093	198.479	70.151	0	0	0.648	19.109	4.32	0.00	4316.66	0	110.318	0	3.108	24.061	22.716
2021/06/28	0.151	119.532	47.172	0	0.223	0.406	8.317	1.88	0.07	1946.68	0	54.775	0.304	3.862	14.882	19.699
2021/07/07	0	31.087	5.579	0	0	0.076	1.783	0.40	0.00	402.77	0	13.397	0	0.452	3.058	9.094
2021/08/02	0	26.924	5.049	0	0	0.063	0.927	0.21	0.00	209.41	0	12.623	0	0.345	2.616	7.055
2021/08/31	0.116	25.131	4.628	0	0	0.056	0.964	0.22	0.00	217.76	0	11.652	0	0.433	2.118	5.879
2021/09/22	0	27.572	4.927	0	0	0.076	0.611	0.14	0.00	138.02	0	14.719	0	0.386	1.737	3.005
Site 4																
2020/10/15	0	39.213	10.038	0	0	0.095	0.965	0.22	0.00	217.99	0	20.893	0	0.669	4.387	4.863
2020/11/26	0.058	76.347	19.978	0	0	0.246	2.358	0.53	0.00	532.66	0	38.959	0.019	1.11	7.34	5.349
2020/12/15	0.07	88.894	20.256	0	0	0.308	1.444	0.33	0.00	326.19	0	44.248	0.036	1.225	7.952	6.182
2021/01/15	0.254	137.053	68.224	0	0	0.453	0.134	0.03	0.00	30.27	0.018	83.614	0.07	3.456	16.383	17.31
2021/02/17	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/03/16	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/04/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/05/11	0	80.488	21.279	0	0	0.195	6.869	1.55	0.00	1551.68	0	40.545	0	0.89	9.392	8.221
2021/05/26	0	43.234	9.099	0	0	0.088	2.828	0.64	0.00	638.84	0	22.604	0	0.554	4.473	3.437
2021/06/28	0.144	116.93	48.95	0	0	0.344	17.405	3.93	0.00	3931.73	0	58.176	0.142	4.013	14.632	13.444
2021/07/07	0	37.953	10.036	0	0	0.087	2.745	0.62	0.00	620.09	0	18.45	0	0.547	4.548	5.197

2021/08/02	0	41.369	13.805	0	0	0.099	2.267	0.51	0.00	512.11	0	21.459	0	0.679	5.203	4.034
2021/08/31	0.066	30.517	8.391	0	0	0.069	1.498	0.34	0.00	338.39	0	15.972	0	0.454	3.247	2.906
2021/09/22	0.049	49.094	18.452	0	0	0.141	1.78	0.40	0.00	402.10	0	27.286	0	0.601	6.225	4.498
Site 5																
2020/10/15	0.073	38.779	8.48	0	0	0.104	0.173	0.04	0.00	39.08	0	20.643	0	0.718	4.019	4.14
2020/11/26	0.055	46.029	10.381	0	0	0.17	0.183	0.04	0.00	41.34	0	24.405	0.024	1.623	4.183	3.306
2020/12/15	0	57.094	14.17	0	0	0.234	0.206	0.05	0.00	46.53	0	28.042	0.053	0.699	5.445	3.77
2021/01/15	0.056	96.259	19.311	0	0	0.4	0.275	0.06	0.00	62.12	0	47.731	0.15	1.211	8.01	5.859
2021/02/17	0	154.435	19.124	0	0	0.482	0.254	0.06	0.00	57.38	0	76.512	0	2.723	13.068	9.178
2021/03/16	0.374	352.371	20.898	0	0	1.414	0	0.00	0.00	0.00	0	176.7	0	10.075	36.376	31.984
2021/04/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/05/11	0	64.645	14.464	0	0	0.151	0.518	0.12	0.00	117.01	0	33.445	0	0.994	6.124	5.079
2021/05/26	0	40.787	8.129	0	0	0.082	0.168	0.04	0.00	37.95	0	21.853	0	0.458	3.177	2.433
2021/06/28	0.087	52.147	11.41	0	0	0.134	1.18	0.27	0.00	266.56	0	24.909	0	0.765	5.05	6.611
2021/07/07	0	40.757	8	0	0	0.093	0.625	0.14	0.00	141.19	0	21.184	0	0.727	5.237	5.902
2021/08/02	0	39.927	9.585	0	0	0.099	0.521	0.12	0.00	117.69	0	20.935	0	0.505	4.253	4.083
2021/08/31	0.09	32.908	6.094	0	0	0.083	0.242	0.05	0.00	54.67	0	17.168	0	0.538	3.231	2.664
2021/09/22	0	38.271	7.476	0	0	0.119	0.292	0.07	0.00	65.96	0	21.299	0	0.372	3.718	2.585
Site 6																
2020/10/15	0	76.602	16.658	0	0	0.227	1.159	0.26	0.00	261.81	0	41.271	0	0.88	8.441	7.466
2020/11/26	0.171	155.295	31.303	0	0	0.522	1.368	0.31	0.00	309.03	0	82.852	0	1.901	16.01	10.967
2020/12/15	0.12	175.929	33.133	0	0	0.595	1.398	0.32	0.00	315.80	0	90.357	0	2.084	16.76	10.603
2021/01/15	0.145	194.802	29.955	0	0	0.619	0.335	0.08	0.00	75.68	0.018	97.147	0	2.956	17.021	11.31
2021/02/17	0.157	212.226	18.362	0	0	0.657	0.316	0.07	0.00	71.38	0	107.712	0	3.132	19.703	13.046
2021/03/16	0.228	302.391	61.499	0	0	1.039	1.953	0.44	0.00	441.18	0	161.291	0	5.565	34.747	26.183
2021/04/15	0.203	282.305	43.865	0	0	0.942	0.982	0.22	0.00	221.83	0	144.258	0	3.392	28.27	20.623
2021/05/11	0.102	187.491	52.051	0	0	0.518	11.1	2.51	0.00	2507.45	0	97.484	0	1.946	20.927	17.338
2021/05/26	0.054	98.131	22.319	0	0	0.249	4.516	1.02	0.00	1020.15	0	51.225	0.025	1.011	10.297	7.639
2021/06/28	0.082	153.02	33.097	0	0	0.436	3.678	0.83	0.00	830.85	0	67.203	0	1.555	14.129	17.788
2021/07/07	0.055	92.731	23.85	0	0	0.244	5.792	1.31	0.00	1308.39	0	45.059	0	0.985	10.884	8.151
2021/08/02	0.065	98.683	25.555	0	0	0.277	3.879	0.88	0.00	876.25	0	51.709	0	1.231	11.26	8.4
2021/08/31	0.06	76.955	20.473	0	0	0.21	3.574	0.81	0.00	807.35	0	37.353	0	0.804	8.625	6.444

2021/09/22	0.089	117.905	27.248	0	0	0.348	2.782	0.63	0.00	628.44	0	59.842	0	1.074	13.216	9.616
Site 7																
2020/10/15	0.269	334.187	47.255	0	0	1.123	1.02	0.23	0.00	230.41	0	181.47	0	3.032	32.055	19.665
2020/11/26	0.247	191.972	17.586	0	0	0.612	0.29	0.07	0.00	65.51	0	103.229	0	2.583	15.831	9.626
2020/12/15	0.162	182.062	15.397	0	0	0.583	0.258	0.06	0.00	58.28	0	92.807	0	2.597	14.988	8.292
2021/01/15	0.184	170.194	12.675	0	0	0.556	0.266	0.06	0.00	60.09	0.035	86.591	0	2.562	13.671	8.52
2021/02/17	0	169.474	11.071	0	0	0.492	0.248	0.06	0.00	56.02	0	87.268	0	2.545	15.634	16.194
2021/03/16	0	165.386	12.549	0	0	0.535	0.34	0.08	0.00	76.80	0	82.792	0	2.51	14.74	9.247
2021/04/15	0.144	162.931	12.443	0	0	0.526	0.356	0.08	0.00	80.42	0	82.004	0	2.411	15.177	10.698
2021/05/11	0.193	310.786	46.061	0	0.201	0.948	25.548	5.77	0.06	5832.40	0	166.969	0	4.008	34.162	25.84
2021/05/26	0.246	345.303	53.171	0	0	1.047	55.793	12.60	0.00	12603.45	0	195.53	0	4.081	46.149	41.219
2021/06/28	0.233	359.927	42.937	0	0	1.136	15.304	3.46	0.00	3457.12	0	195.874	0	3.955	40.468	33.43
2021/07/07	0.209	263.879	50.715	0	0	0.785	40.201	9.08	0.00	9081.27	0	152.315	0	2.888	34.978	29.681
2021/08/02	0.232	385.514	65.678	0	0	1.223	14.883	3.36	0.00	3362.02	0	209.598	0	2.953	46.732	34.872
2021/08/31	0.226	286.279	48.415	0	0	0.842	9.474	2.14	0.00	2140.14	0	158.798	0	2.049	32.393	25.299
2021/09/22	0.311	451.16	64.953	0	0	1.466	1.769	0.40	0.00	399.61	0	228.922	0	3.124	49.965	33.258
Site 8																
2020/10/15	0.062	84.453	19.318	0	0	0.256	1.171	0.26	0.00	264.52	0	47.706	0	0.996	10.325	8.878
2020/11/26	0.109	154.61	29.434	0	0	0.519	1.254	0.28	0.00	283.27	0	82.303	0	1.742	15.594	10.554
2020/12/15	0.122	173.762	29.818	0	0	0.584	0.671	0.15	0.00	151.58	0	85.634	0	1.939	16.53	10.146
2021/01/15	0.124	157.078	18.487	0	0	0.507	0.05	0.01	0.00	11.29	0	79.652	0	2.151	13.612	8.338
2021/02/17	0.104	162.792	12.401	0	0	0.501	0	0.00	0.00	0.00	0	83.566	0	2.68	15.226	11.678
2021/03/16	0.175	238.009	37.499	0	0	0.806	0.612	0.14	0.00	138.25	0	123.923	0	3.804	24.892	17.825
2021/04/15	0.153	206.149	26.344	0	0	0.68	0.254	0.06	0.00	57.38	0	104.531	0	2.44	20.106	13.708
2021/05/11	0.122	235.119	53.833	0	0	0.653	15.042	3.40	0.00	3397.94	0	126.293	0.055	2.934	27.337	22.709
2021/05/26	0.069	123.185	25.42	0	0	0.332	7.61	1.72	0.00	1719.07	0	66.363	0.102	1.409	14.271	11.598
2021/06/28	0.093	160.358	32.104	0	0	0.46	3.94	0.89	0.00	890.03	0	73.142	0	1.896	16.986	14.957
2021/07/07	0	115.825	27.432	0	0	0.33	7.577	1.71	0.00	1711.62	0	63.379	0	1.498	17.855	16.136
2021/08/02	0.074	116.133	27.355	0	0	0.329	4.078	0.92	0.00	921.21	0	57.632	0	0.903	14.096	12.021
2021/08/31	0.106	99.035	22.403	0	0	0.264	3.746	0.85	0.00	846.21	0	47.026	0	0.88	10.94	9.307
2021/09/22	0.076	134.305	28.119	0	0	0.388	2.652	0.60	0.00	599.08	0	66.347	0	1.122	15.122	11.471
Site 9																

2020/10/15	0.119	92.667	20.727	0	0	0.282	1.165	0.26	0.00	263.17	0	49.24	0	1.166	9.818	8.283
2020/11/26	0.109	161.418	28.8	0	0	0.526	0.898	0.20	0.00	202.86	0	84.358	0	1.837	15.794	10.62
2020/12/15	0.133	207.438	33.287	0	0	0.69	0.395	0.09	0.00	89.23	0.017	103.931	0	2.129	19.37	12.599
2021/01/15	0.16	212.201	23.537	0	0	0.686	0.248	0.06	0.00	56.02	0	108.775	0	2.589	18.317	11.872
2021/02/17	0.152	214.39	15.893	0	0	0.647	0.205	0.05	0.00	46.31	0	109.638	0	2.756	19.513	14.993
2021/03/16	0.173	229.018	26.929	0	0	0.762	0.245	0.06	0.00	55.34	0	117.8	0	3.109	23.004	16.122
2021/04/15	0.169	218.039	22.918	0	0	0.709	0.423	0.10	0.00	95.55	0	110.707	0	2.486	20.806	14.234
2021/05/11	0.106	210.312	43.451	0	0	0.565	11.78	2.66	0.00	2661.06	0	107.472	0	2.642	21.495	17.095
2021/05/26	0.083	147.866	28.645	0	0	0.398	8.509	1.92	0.00	1922.15	0	78.639	0.122	1.776	16.349	13.265
2021/06/28	0.105	181.423	33.351	0	0	0.516	4.894	1.11	0.00	1105.54	0	81.978	0	2.296	18.052	15.276
2021/07/07	0	131.124	29.96	0	0	0.369	8.084	1.83	0.00	1826.15	0	70.744	0	1.712	19.024	17.748
2021/08/02	0.079	145.742	31.473	0	0	0.41	4.482	1.01	0.00	1012.47	0	70.935	0	1.222	16.464	12.815
2021/08/31	0.099	110.922	24.645	0	0	0.31	3.946	0.89	0.00	891.39	0	55.715	0	1.086	12.333	9.7
2021/09/22	0.083	159.557	31.997	0	0	0.478	2.971	0.67	0.00	671.14	0	81.613	0	1.374	17.962	13.387
Site 10																
2020/10/15	0	38.213	2.667	0	0	0.087	0.056	0.01	0.00	12.65	0	19.838	0	0.858	3.296	3.662
2020/11/26	0	44.752	3.773	0	0	0.144	0.068	0.02	0.00	15.36	0	23.175	0	1.018	2.903	2.278
2020/12/15	0	50.418	4.518	0	0	0.181	0.085	0.02	0.00	19.20	0	24.18	0	1.136	3.744	2.714
2021/01/15	0.065	55.14	5.151	0.183	0	0.23	0.091	0.02	0.00	20.56	0	27.542	0	1.56	3.824	2.828
2021/02/17	0.069	93.467	9.857	0	0	0.352	0.243	0.05	0.00	54.89	0	48.066	0	3.136	5.586	6.514
2021/03/16	0.078	108.759	13.645	0	0	0.411	0.468	0.11	0.00	105.72	0	57.859	0	3.716	5.919	3.03
2021/04/15	0.08	108.668	14.099	0	0	0.411	0.519	0.12	0.00	117.24	0	56.86	0	3.171	4.927	2.407
2021/05/11	0.048	53.186	5.831	0	0	0.14	0.289	0.07	0.00	65.28	0	26.182	0	1.331	3.332	6.116
2021/05/26	0	36.449	3.991	0	0	0.064	0.124	0.03	0.00	28.01	0	18.428	0	0.705	2.335	4.056
2021/06/28	0	38.455	3.887	0	0	0.085	0.25	0.06	0.00	56.47	0	15.506	0	0.918	2.925	8.667
2021/07/07	0.082	33.291	3.87	0	0	0.064	0.231	0.05	0.00	52.18	0	15.465	0	0.648	3.389	5.181
2021/08/02	0	34.765	4.497	0	0	0.083	0.206	0.05	0.00	46.53	0	18.58	0	0.797	3.274	4.617
2021/08/31	0	26.341	3.164	0	0	0.054	0.248	0.06	0.00	56.02	0	13.497	0	0.494	2.571	3.829
2021/09/22	0	35.609	3.975	0	0	0.117	0.182	0.04	0.00	41.11	0	19.285	0	0.664	2.686	2.877
Site 11																
2020/10/15	0.076	92.54	20.723	0	0	0.286	1.085	0.25	0.00	245.10	0	49.778	0	1.189	9.673	7.837
2020/11/26	0.109	149.895	27.329	0	0	0.492	0.742	0.17	0.00	167.62	0	78.362	0	1.945	13.584	9.128

2020/12/15	0.132	203.48	33.143	0	0	0.666	0.333	0.08	0.00	75.22	0.016	102.606	0	2.547	17.899	10.932
2021/01/15	0.14	176.649	23.324	0	0	0.582	0.215	0.05	0.00	48.57	0	90.516	0	2.611	13.973	8.644
2021/02/17	0.141	172.202	17.456	0	0	0.525	0.161	0.04	0.00	36.37	0	89.655	0	3.061	15.671	13.49
2021/03/16	0	187.858	20.502	0	0	0.629	0.344	0.08	0.00	77.71	0	95.631	0	3.197	16.839	10.429
2021/04/15	0.147	176.181	19.618	0	0	0.574	0.377	0.09	0.00	85.16	0	88.301	0	2.548	15.365	10.323
2021/05/11	0.13	226.866	46.193	0	0	0.61	12.339	2.79	0.00	2787.34	0	121.046	0	3.241	24.492	20.671
2021/05/26	0.087	148.328	29.376	0	0	0.404	7.622	1.72	0.00	1721.78	0	80.653	0.083	1.937	15.53	12.487
2021/06/28	0.102	180.422	32.1	0	0	0.522	3.316	0.75	0.00	749.07	0	80.714	0	2.531	14.401	15.149
2021/07/07	0	137.445	31.94	0	0	0.384	7.76	1.75	0.00	1752.96	0	75.666	0	1.777	17.742	15.673
2021/08/02	0.081	152.287	33.514	0	0	0.433	3.974	0.90	0.00	897.71	0	77.804	0	1.262	17.858	14.104
2021/08/31	0.088	114.818	24.769	0	0	0.322	3.432	0.78	0.00	775.28	0	57.475	0	1.044	12.432	9.805
2021/09/22	0.082	165.91	31.319	0	0	0.465	2.529	0.57	0.00	571.29	0	79.03	0	1.319	16.978	12.769
Site 12																
2020/10/15	0	58.418	8.441	0	0	0.177	3.026	0.68	0.00	683.56	0	30.705	0	0.898	5.831	5.288
2020/11/26	0	70.955	8.997	0	0	0.256	2.892	0.65	0.00	653.29	0	36.49	0	1.029	5.76	3.834
2020/12/15	0	70.362	8.181	0	0	0.271	1.583	0.36	0.00	357.59	0.017	33.379	0	0.922	5.853	3.842
2021/01/15	0	62.815	6.816	0	0	0.232	1.026	0.23	0.00	231.77	0.017	31.534	0	0.948	2.824	2.049
2021/02/17	0	59.909	5.101	0	0	0.187	1.498	0.34	0.00	338.39	0	33.255	0	1.3	5.188	5.701
2021/03/16	0	66.619	7.432	0	0	0.232	1.231	0.28	0.00	278.08	0	33.902	0	1.381	5.093	5.157
2021/04/15	0	66.971	7.535	0	0	0.237	1.343	0.30	0.00	303.38	0	33.406	0	0.965	4.301	5.637
2021/05/11	0	84.413	12.561	0	0	0.215	1.547	0.35	0.00	349.46	0	41.475	0	1.481	7.515	5.81
2021/05/26	0	57.87	8.063	0	0	0.142	1.068	0.24	0.00	241.26	0	29.162	0	0.741	5.043	3.667
2021/06/28	0	58.639	6.474	0	0	0.158	1.092	0.25	0.00	246.68	0	25.477	0	0.779	5.813	6.17
2021/07/07	0	54.617	8.076	0	0	0.138	1.571	0.35	0.00	354.88	0	25.286	0	0.637	6.45	5.151
2021/08/02	0	60.368	9.815	0	0	0.165	3.48	0.79	0.00	786.12	0	28.488	0	0.92	6.355	5.03
2021/08/31	0.086	58.417	8.791	0	0	0.145	1.687	0.38	0.00	381.09	0	26.444	0	0.952	5.51	3.744
2021/09/22	0	58.149	8.411	0	0	0.185	2.119	0.48	0.00	478.68	0	30.352	0	0.848	6.093	4.379
Site 13																
2020/10/15	0.061	84.309	17.077	0	0	0.259	1.722	0.39	0.00	388.99	0	44.892	0	1.127	8.412	7.034
2020/11/26	0.087	115.638	19.77	0	0	0.386	1.743	0.39	0.00	393.74	0	60.849	0	1.674	10.076	6.497
2020/12/15	0.08	128.732	19.793	0	0	0.441	1.139	0.26	0.00	257.30	0	63.854	0	1.534	10.671	6.184
2021/01/15	0.106	132.044	19.131	0	0	0.436	1.103	0.25	0.00	249.16	0	67.058	0	1.819	9.174	5.142

2021/02/17	0.088	128.503	14.907	0	0	0.379	1.419	0.32	0.00	320.55	0	67.039	0	1.871	11.681	10.355
2021/03/16	0	137.63	16.807	0	0	0.458	0.966	0.22	0.00	218.22	0	70.654	0	2.562	12.189	8.375
2021/04/15	0.095	122.095	15.457	0	0	0.401	1.172	0.26	0.00	264.75	0	61.565	0	1.719	10.788	7.807
2021/05/11	0.113	216.657	42.737	0	0	0.575	10.85	2.45	0.00	2450.98	0	111.534	0	2.813	21.016	16.815
2021/05/26	0.068	120.049	22.456	0	0	0.317	5.022	1.13	0.00	1134.45	0	64.802	0.063	1.581	12.148	9.547
2021/06/28	0.08	125.938	21.593	0	0	0.358	2.81	0.63	0.00	634.77	0	56.324	0	1.98	11.587	12.037
2021/07/07	0	121.164	27.184	0	0	0.336	6.414	1.45	0.00	1448.90	0	67.644	0	1.581	15.957	14.279
2021/08/02	0.07	133.543	28.545	0	0	0.375	3.942	0.89	0.00	890.48	0	67.144	0	1.21	14.897	12.169
2021/08/31	0.11	101.933	21.681	0	0	0.288	2.985	0.67	0.00	674.30	0	52.607	0	1.031	11.077	8.958
2021/09/22	0.067	130.741	25.206	0	0	0.381	2.47	0.56	0.00	557.96	0	65.497	0	1.206	13.623	9.892
Site 14																
2020/10/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2020/11/26	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2020/12/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/01/15	7.117	648.548	35.969	0	0	2.325	0.101	0.02	0.00	22.82	0.167	366.935	1.406	47.793	51.384	32.41
2021/02/17	0.427	888.923	16.946	0.423	0	3.251	0	0.00	0.00	0.00	0	506.312	0	48.639	72.629	43.338
2021/03/16	0.467	1310.399	24.944	0	0	4.679	0	0.00	0.00	0.00	0	725.806	0	49.608	101.505	52.462
2021/04/15	0.528	2246.245	84.41	0	0	8.001	0	0.00	0.00	0.00	0	1231.851	0	48.85	168.693	84.007
2021/05/11	0.201	345.78	63.571	0.806	0.273	1.045	31.287	7.07	0.08	7150.75	0	196.488	0.092	14.832	33.756	31.917
2021/05/26	0.221	400.808	64.343	0.527	0	1.272	27.919	6.31	0.00	6306.81	0	232.041	0	13.409	40.133	36.809
2021/06/28	0.225	622.64	78.588	1.467	0	2.041	6.949	1.57	0.00	1569.76	0	345.876	0	33.028	57.252	45.802
2021/07/07	0.236	436.673	72.044	0.573	0	1.459	29.449	6.65	0.00	6652.43	0	265.015	0	13.501	43.211	34.906
2021/08/02	0.224	458.122	73.009	0.318	0	1.534	11.946	2.70	0.00	2698.56	0	267.828	0	12.396	44.287	34.692
2021/08/31	0.213	263.881	44.475	0	0	0.861	8.546	1.93	0.00	1930.51	0	162.606	0	7.438	27.293	23.317
2021/09/22	0.248	545.684	69.088	0.644	0	1.803	9.028	2.04	0.00	2039.40	0	299.072	0	19.591	48.102	33.949
Site 15																
2020/10/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2020/11/26	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2020/12/15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2021/01/15	0.219	251.485	6.676	0	0	0.986	0	0.00	0.00	0.00	0	137.868	0	7.649	22.881	15.621
2021/02/17	0.203	294.773	4.965	0	0	1.122	0	0.00	0.00	0.00	0	164.101	0	8.201	26.126	19.133
2021/03/16	0.274	381.391	13.725	0	0	1.548	0	0.00	0.00	0.00	0	210.305	0	11.869	35.004	21.897

2021/04/15	0.195	259.213	6.338	0	0	0.995	0	0.00	0.00	0.00	0	138.559	0	4.91	21.303	14.512
2021/05/11	0.172	290.72	54.652	0.426	0.281	0.865	26.124	5.90	0.09	5986.88	0	164.096	0.173	12.388	28.788	28.129
2021/05/26	0.18	309.368	50.605	0	0	0.949	17.425	3.94	0.00	3936.25	0	178.585	0	9.311	28.901	26.754
2021/06/28	0.132	275.526	35.496	0	0	0.874	1.854	0.42	0.00	418.81	0	150.283	0	10.958	27.211	22.884
2021/07/07	0.149	257.332	47.346	0	0	0.816	15.447	3.49	0.00	3489.43	0	152.019	0	6.682	27.017	24.142
2021/08/02	0.161	270.862	46.392	0	0	0.895	3.671	0.83	0.00	829.27	0	156.772	0	6.676	26.985	25.787
2021/08/31	0.217	194.541	33.748	0	0	0.578	5.507	1.24	0.00	1244.01	0	101.92	0	3.897	15.377	15.522
2021/09/22	0.135	296.707	41.56	0	0	0.931	2.823	0.64	0.00	637.71	0	153.91	0	6.994	28.691	22.526

Table B2 Stable isotope composition for collected samples at each Bot River site during the year.

Date	Sample site	^{2}H	^{18}O	Type	D-excess against GMWL	D-excess against LMWL
2020/10/14	S1	-7.8	-1.73	River	-4.5351	-11.06054
2020/10/14	S3	-16.4	-3.77	River	3.4501	-1.90846
2020/10/14	S4	-21.6	-4.65	River	5.4045	0.5493
2020/10/14	S5	-18.5	-4.66	River	8.5858	3.73632
2020/10/14	S6T	-17.1	-3.94	River	4.1322	-1.12912
2020/10/14	S6B	-16.9	-3.91	River	4.0883	-1.19018
2020/10/14	S7	-26.2	-5.29	River	6.0077	1.51858
2020/10/14	S8	-17.6	-4.01	River	4.2013	-1.01998
2020/10/14	S9	-17.8	-4.04	River	4.2452	-0.95892
2020/10/14	S10	-21.3	-5.02	River	8.7126	4.06904
2020/10/14	S11T	-18.2	-4.02	River	3.6826	-1.53296
2020/10/14	S11B	-18.3	-4.1	River	4.233	-0.9368
2020/10/14	S12	-20.3	-4.79	River	7.8427	3.06758
2020/10/14	S13T	-19.1	-4.46	River	6.3598	1.39592
2020/10/14	S13B	-19	-4.43	River	6.2159	1.23486
2021/03/16	S1	0.2	-0.31	River	-8.0797	-15.41738
2021/03/16	S3	-18.5	-4.08	River	3.8704	-1.31084
2021/03/16	S5	-20.6	-4.01	River	1.2013	-4.01998
2021/03/16	S6	-9.2	-1.88	River	-4.7156	-11.15524
2021/03/16	S7	-28.3	-5.5	River	5.615	1.246
2021/03/16	S8	-14.9	-3.05	River	-0.9035	-6.6739
2021/03/16	S9	-17.8	-3.6	River	0.668	-4.7878
2021/03/16	S10	-25.1	-4.83	River	3.3679	-1.38434
2021/03/16	S11	-21	-4.22	River	2.5086	-2.59256
2021/03/16	S12	-25.4	-5.25	River	6.4825	1.9705
2021/03/16	S13	-22.8	-4.6	River	3.798	-1.0858
2021/03/16	S14	4.2	0.44	River	-10.1772	-17.94388
2021/03/16	S15	-10.8	-2.38	River	-2.2506	-8.40424
2021/03/16	PPT_03	6.6	-1.55	Precipitation	8.4015	1.7731
2021/04/14	S1	0.1	-0.05	River	-10.2935	-17.7799
2021/04/14	S6	-9.5	-2.06	River	-3.5522	-9.88888
2021/04/14	S7	-26.4	-5.08	River	4.1004	-0.50884
2021/04/14	S8	-18.1	-3.62	River	0.5306	-4.91376
2021/04/14	S9	-15.9	-3.6	River	2.568	-2.8878
2021/04/14	S10	-23.4	-4.48	River	2.2224	-2.73004
2021/04/14	S11	-19	-3.93	River	2.1509	-3.11614
2021/04/14	S12	-23.2	-4.81	River	5.1053	0.34162
2021/04/14	S13	-23.5	-4.77	River	4.4801	-0.30646
2021/04/14	S15	-15.3	-3.34	River	1.0542	-4.55032
2021/04/14	PPT_04	3.6	-2.18	Precipitation	10.5234	4.25536
2021/04/29	PPT 29/04	-14.1	-3.07	Precipitation	0.0591	-5.69986
2021/05/06	PPT 06/05	-29.9	-6.3	Precipitation	10.519	6.6076
2021/05/05	PPT 05/05	-40.7	-6.58	Precipitation	1.9954	-1.75584
2021/05/11	S10	-22	-4.48	River	3.6224	-1.33004
2021/05/11	S5	-23.3	-4.11	River	-0.6857	-5.84978
2021/05/11	S12	-23.6	-4.43	River	1.6159	-3.36514

2021/05/11	S4	-25.9	-4.64	River	1.0232	-3.83772
2021/05/11	S6	-20.4	-3.54	River	-2.4198	-7.90992
2021/05/11	S9	-23.2	-4.1	River	-0.667	-5.8368
2021/05/11	S13	-25.9	-4.75	River	1.9175	-2.8805
2021/05/11	S11	-23.8	-5.27	River	8.2451	3.74454
2021/05/11	S8	-24.2	-5.77	River	11.9101	7.69554
2021/05/11	S3	-12.7	-2.73	River	-1.3051	-7.25854
2021/05/11	S15	-21.1	-3.79	River	-1.0873	-6.43442
2021/05/11	S7	-23	-3.87	River	-2.3369	-7.63826
2021/05/11	S14	-19.7	-3.51	River	-1.9637	-7.47098
2021/05/11	S1	-9.3	-1.59	River	-7.1733	-13.77882
2021/06/28	S1	-10.7	-2.16	River	-3.9392	-10.21868
2021/06/28	S2	-10.4	-1.85	River	-6.1595	-12.6163
2021/06/28	S3	-14	-3.27	River	1.7851	-3.85946
2021/06/28	S4	-14.9	-3.32	River	1.2916	-4.32436
2021/06/28	S5	-23.5	-4.63	River	3.3419	-1.52474
2021/06/28	S6T	-16.5	-3.75	River	3.1875	-2.1825
2021/06/28	S7	-12.3	-2.76	River	-0.6612	-6.59748
2021/06/28	S8	-18	-3.97	River	3.4761	-1.76806
2021/06/28	S9	-18.3	-3.96	River	3.0948	-2.15508
2021/06/28	S10	-24.6	-5.01	River	5.3313	0.68202
2021/06/28	S11	-19.5	-3.77	River	0.3501	-5.00846
2021/06/28	S12	-20.6	-4.3	River	3.559	-1.4964
2021/06/28	S13	-21.3	-4.23	River	2.2899	-2.80554
2021/06/28	E7	-14.5	-3.2	River	0.716	-4.9686
2021/06/28	S15	-18.2	-3.84	River	2.2192	-3.09932
2021/07/12	PPT 12/07	-26.8	-4.711	Precipitation	0.70043	-4.119878
2021/07/21	PPT 21/07	-25.36	-4.897	Precipitation	3.65261	-1.061306
2021/08/02	S1	-11.54	-2.212	River	-4.35644	-10.606176
2021/08/02	S2	-19.05	-3.493	River	-1.45191	-6.968914
2021/08/02	S3	-27.82	-5.244	River	4.01372	-0.501712
2021/08/02	S4	-28.17	-5.36	River	4.6068	0.15772
2021/08/02	S5	-26.49	-5.487	River	7.31931	2.942874
2021/08/02	S6	-23.35	-4.784	River	4.74392	-0.034632
2021/08/02	S7	-17.74	-3.839	River	2.67107	-2.648022
2021/08/02	S8	-23.29	-4.73	River	4.3649	-0.44454
2021/08/02	S9	-21.53	-4.243	River	2.16559	-2.922414
2021/08/02	S10	-26.48	-5.379	River	6.45127	2.013058
2021/08/02	S11	-23.21	-4.506	River	2.62378	-2.313788
2021/08/02	S12	-23.69	-4.92	River	5.5096	0.80884
2021/08/02	S13	-23.69	-4.703	River	3.74539	-1.079494
2021/08/02	S14	-19.71	-4.203	River	3.66039	-1.450494
2021/08/02	S15	-21.68	-4.365	River	3.00745	-2.01077
2021/08/31	S1	-10.3	-2.288	River	-2.49856	-8.704824
2021/08/31	S2	-17.12	-3.291	River	-1.16417	-6.796718
2021/08/31	S3	-26.97	-5.331	River	5.57103	1.105362
2021/08/31	S4	-27.78	-5.597	River	6.92361	2.610094
2021/08/31	S5	-26.93	-5.67	River	8.3671	4.09534
2021/08/31	S6	-23.8	-5.276	River	8.29388	3.796752

2021/08/31	S7	-19.2	-3.897	River	1.68261	-3.603306
2021/08/31	S8	-25.7	-4.89	River	3.2557	-1.46222
2021/08/31	S9	-24.61	-5.027	River	5.45951	0.819954
2021/08/31	S10	-29.85	-5.629	River	5.11377	0.818558
2021/08/31	S11	-25.08	-5.137	River	5.88381	1.307174
2021/08/31	S12	-22.93	-5.39	River	10.0907	5.65878
2021/08/31	S13	-23.16	-4.733	River	4.51929	-0.288434
2021/08/31	S14	-24.35	-4.913	River	4.79269	0.087926
2021/08/31	S15	-25.73	-5.059	River	4.59967	-0.021582
2021/09/22	S1	-11.04	-2.184	River	-4.08408	-10.349832
2021/09/22	S3	-25.25	-5.103	River	5.43739	0.841306
2021/09/22	S4	-25.06	-4.986	River	4.67618	0.013172
2021/09/22	S5	-26.75	-5.363	River	6.05119	1.603826
2021/09/22	S6	-24.27	-4.528	River	1.74264	-3.182344
2021/09/22	S7	-20.75	-4.145	River	2.14885	-2.99521
2021/09/22	S8	-20.23	-4.437	River	5.04281	0.065774
2021/09/22	S9	-21.51	-4.481	River	4.12053	-0.831338
2021/09/22	S10	-25.28	-5.481	River	8.48053	4.100662
2021/09/22	S11	-20.24	-4.605	River	6.39865	1.51771
2021/09/22	S12	-23.84	-5.143	River	7.17259	2.599386
2021/09/22	S13	-25.07	-4.896	River	3.93448	-0.780008
2021/09/22	S14	-17.29	-4.052	River	4.85276	-0.344496
2021/09/22	S15	-20.79	-4.676	River	6.42588	1.585552

APPENDIX C: BOT RIVER SOIL ANALYSIS

Table C1 Samples collected for classification of soil form at each transect.

Transect	Site	Horizon	Date	Latitude	Longitude	Diagnostic horizons	Lower depth (mm)	Colour (Munsell code)	Soil form
1	1	A	07-May-19	34°10'23.4"S	19°15'55.2"E	Orthic A	200	7.5YR6/4	
1	1	B	07-May-19	34°10'23.4"S	19°15'55.2"E	Prismacutanic B	500	7.5YR4/6	
1	1	C	07-May-19	34°10'23.4"S	19°15'55.2"E	Lithocutanic B (Saprolite)	1500+		
1	1					Saprolite			Ss2200
1	2	A	07-May-19	34°10'23.5"S	19°15'56.2"E	Orthic A	150	7.5YR6/4	
1	2	B	07-May-19	34°10'23.5"S	19°15'56.2"E	Pedocutanic B (Neocutanic B)	500	7.5YR5/4	
1	2	C	07-May-19	34°10'23.5"S	19°15'56.2"E	Lithocutanic B (Saprolite)	900+		
1	2					Saprolite			Sw1/2111
1	3	A	07-May-19	34°10'23.4"S	19°15'56.4"E	Orthic A	100	7.5YR6/4	
1	3	B1	07-May-19	34°10'23.4"S	19°15'56.4"E	Pedocutanic B	400	7.5YR5/4	
1	3	B2	07-May-19	34°10'23.4"S	19°15'56.4"E	Lithocutanic B	900+	7.5YR4/6	
1	3	C				Saprolite			Sw2110
2	1	A	07-May-19	34°10'30.3"S	19°15'40.7"E	Orthic A	200	10YR7/2	
2	1	E	07-May-19	34°10'30.3"S	19°15'40.7"E	E horizon	400	10YR7/2	
2	1	B	07-May-19	34°10'30.3"S	19°15'40.7"E	Pedocutanic B	550	10YR4/6	
2	1	C				Saprolite	550+		Km1120
2	2	A	07-May-19	34°10'30.30"S	19°15'40.10"E	Orthic A	150	10YR7/2	
2	2	E	07-May-19	34°10'30.30"S	19°15'40.10"E	E horizon	450	10YR7/3	
2	2	B	07-May-19	34°10'30.30"S	19°15'40.10"E	Pedocutanic B	900	10YR5/6	
2	2	C				Saprolite	900+		Km111/20
2	3	A	07-May-19	34°10'30.20"S	19°15'39.60"E	Orthic A	100	10YR7/3	
2	3	B1	07-May-19	34°10'30.20"S	19°15'39.60"E	E horizon	350	10YR7/3	
2	3	B2	07-May-19	34°10'30.20"S	19°15'39.60"E	Pedocutanic B	600	10YR6/8	
2	3	C				Saprolite	600+		Km1120
3	1	A	07-May-19	34°10'12.8"S	19°15'29.1"E	Orthic A	200	10YR6/2	Ss2100

3	1	B	07-May-19	34°10'12.8"S	19°15'29.1"E	Prismacutanic B	400	10YR3/2	
3	1	C				Saprolite	400+		
3	2	A	07-May-19	34°10'13.3"S	19°15'29.0"E	Orthic A	100	10YR7/2	
3	2	B1	07-May-19	34°10'13.3"S	19°15'29.0"E	Neocutanic B/E horizon	300	10YR6/2	
3	2	B2	07-May-19	34°10'13.3"S	19°15'29.0"E	Pedocutanic B	500	10YR3/3	
3	2	C				Saprolite	500+		Tu212
3	3	A	07-May-19	34°10'14.0"S	19°15'28.9"E	Orthic A	300	10YR6/2	
3	3	B	07-May-19	34°10'14.0"S	19°15'28.9"E	Pedocutanic B	400	10YR4/2	
3	3	C				Saprolite	400+		Sw2121

Table D2 Description of soil texture for each sample site.

Transect	Site	Horizon	% soil >2mm	% soil <2mm	% organics in soil	Mass (g) coarse sand (>0.5mm)	Mass (g) medium sand (>0.25mm)	Mass (g) fine sand (>0.106mm)	Mass (g) very fine sand (>0.053mm)	Mass (g) extra silt and clay (<0.053mm)	Mass (g) of sand fraction	Sand fraction (%)	% sequioxidic material removed	Clay (%)	Fine silt (%)	Coarse silt (%)	Soil texture class
1	1	A	55.21	44.79	2.25	4.10	1.70	2.40	0.90	0.10	9.10	23.27	5.12	34.73	20.41	21.59	Clay Loam
1	1	B	59.39	40.61	3.25	0.30	0.30	0.50	0.10	0.10	1.20	3.10	10.85	51.10	30.97	14.82	Silty Clay
1	1	C	91.02	8.98	1.75	13.20	3.30	2.30	0.50	0.10	19.30	49.11	1.78	21.04	20.36	9.49	Loam
1	1																
1	2	A	59.04	40.96	2.50	4.70	1.80	2.50	1.30	0.20	10.30	26.41	6.67	36.59	10.00	27.00	Clay Loam
1	2	B	68.31	31.69	1.75	3.00	1.30	2.30	0.40	0.00	7.00	17.81	6.36	35.67	30.53	15.98	Silty Clay Loam
1	2	C	65.44	34.56	3.25	1.70	0.80	1.00	0.30	0.10	3.80	9.82	3.62	54.24	20.56	15.38	Clay
1	2																
1	3	A	55.22	44.78	2.50	4.00	1.80	3.20	2.10	0.80	11.10	28.46	5.13	35.90	20.10	15.54	Clay Loam
1	3	B1	57.20	42.80	2.00	2.20	0.90	1.90	1.00	0.10	6.00	15.31	9.44	49.20	20.32	15.17	Clay
1	3	B2	57.09	42.91	4.00	0.50	0.40	1.00	0.20	0.00	2.10	5.47	9.90	60.73	20.83	12.97	Clay
1	3	C															
2	1	A	69.82	30.18	2.00	4.30	1.30	2.20	0.60	0.30	8.40	21.43	7.40	37.24	20.26	21.07	Clay Loam
2	1	E	78.29	21.71	2.00	7.70	0.90	1.40	0.40	0.00	10.40	26.53	4.85	24.03	20.41	29.03	Silt Loam
2	1	B	81.30	18.70	3.00	3.30	0.60	0.90	0.10	0.10	4.90	12.63	6.70	46.85	30.89	9.63	Silty Clay

2	1	C						0.00	0.00	0.00							
2	2	A	60.25	39.75	2.25	4.50	1.40	2.60	0.90	0.00	9.40	24.04	5.12	24.35	30.69	20.92	Silt Loam
2	2	E	78.38	21.62	1.75	4.40	1.40	2.60	1.20	0.20	9.60	24.43	5.85	45.75	9.87	19.95	Clay
2	2	B	53.95	46.05	2.25	1.00	0.60	1.50	0.30	0.00	3.40	8.70	5.88	55.81	20.46	15.04	Clay
2	2	C															
2	3	A	53.93	46.07	2.25	4.50	1.30	2.40	1.10	0.30	9.30	23.79	5.12	35.04	20.31	20.87	Clay Loam
2	3	B1	80.04	19.96	2.25	4.10	1.10	2.00	0.40	0.00	7.60	19.44	6.65	36.11	30.69	13.76	Silty Clay Loam
2	3	B2	66.10	33.90	2.50	5.20	0.80	1.30	0.30	0.10	7.60	19.49	6.92	46.89	20.43	13.20	Clay
2	3	C						0.00	0.00	0.00							
3	1	A	58.59	41.41	3.00	3.30	1.70	3.50	1.60	0.20	10.10	26.03	4.90	24.48	31.03	18.45	Loam
3	1	B	71.61	28.39	2.75	3.90	1.50	2.50	0.90	0.20	8.80	22.62	4.11	44.30	30.77	2.30	Clay
3	1	C															
3	2	A	56.40	43.60	2.00	4.40	2.10	3.70	1.30	0.20	11.50	29.34	7.14	36.84	20.31	13.52	Clay Loam
3	2	B1	65.82	34.18	1.75	3.40	1.70	3.50	1.30	0.30	9.90	25.19	4.58	34.35	20.20	20.25	Clay Loam
3	2	B2	63.38	36.62	2.75	1.40	0.80	1.80	0.50	0.00	4.50	11.57	5.40	45.30	30.85	12.29	Silty Clay
3	2	C															
3	3	A	57.73	42.27	3.75	2.60	1.70	4.10	1.80	0.30	10.20	26.49	6.49	26.42	20.78	26.31	Loam
3	3	B	63.00	37.00	3.00	2.20	1.70	3.20	0.80	0.20	7.90	20.36	3.61	43.95	20.45	15.24	Clay
		C															

Table C3 Raw data for determination of plant (bio) available phosphorus, ammonium and nitrate in soils at each sample site – Phase 1 of soil sampling.

Transect	Site	Sample	Depth (mm)	Field position	Season	Plant-available phosphorus (ppm)	Plant-available phosphorus (mg/kg)	NH_4 (mg/kg)	NO_3 (mg/kg)
1	1	a	200	Cultivated	Winter	4.88	36.63	106.28	64.35
1	1	b	200	Cultivated	Winter	5.27	39.53	11.48	86.11
1	1	c	200	Cultivated	Winter	3.68	27.62	33.08	42.59
1	1	a	300	Cultivated	Winter	3.17	23.76	33.48	69.44
1	1	b	300	Cultivated	Winter	2.39	17.96	0.00	75.93
1	1	c	300	Cultivated	Winter	3.38	25.36	33.08	19.44

1	2	a	150	Cultivated	Winter	4.84	36.31	20.68	55.56
1	2	b	150	Cultivated	Winter	4.50	33.73	25.08	50.00
1	2	c	150	Cultivated	Winter	6.26	46.93	7.08	49.07
1	2	a	300	Cultivated	Winter	2.05	15.39	0.00	24.07
1	2	b	300	Cultivated	Winter	2.27	17.00	46.68	150.00
1	2	c	300	Cultivated	Winter	2.87	21.50	0.00	34.26
1	3	a	150	Renosterveld	Winter	1.41	10.56	11.48	57.41
1	3	b	150	Renosterveld	Winter	0.59	4.44	0.00	145.37
1	3	c	150	Renosterveld	Winter	0.76	5.73	59.48	39.81
1	4	a	150	Renosterveld	Winter	0.12	0.90	46.68	71.30
1	4	b	150	Renosterveld	Winter	0.08	0.58	1.48	113.89
1	4	c	150	Renosterveld	Winter	0.46	3.48	15.88	60.19
2	1	a	200	Cultivated	Winter	5.53	41.46	41.08	42.59
2	1	b	200	Cultivated	Winter	5.70	42.75	24.68	64.81
2	1	c	200	Cultivated	Winter	4.54	34.06	6.28	80.56
2	2	a	200	Cultivated	Winter	4.93	36.95	21.48	110.19
2	2	b	200	Cultivated	Winter	4.28	32.12	13.48	45.37
2	2	c	200	Cultivated	Winter	3.51	26.33	17.88	44.44
2	3	a	150	Renosterveld	Winter	2.09	15.71	0.00	74.07
2	3	b	150	Renosterveld	Winter	1.84	13.78	57.88	57.41
2	3	c	150	Renosterveld	Winter	2.31	17.32	11.48	41.67
2	4	a	150	Renosterveld	Winter	1.11	8.30	7.48	91.67
2	4	b	150	Renosterveld	Winter	1.84	13.78	0.00	81.48
2	4	c	150	Renosterveld	Winter	1.28	9.59	0.00	35.19
3	1	a	200	Cultivated	Winter	9.18	68.82	17.08	115.74
3	1	b	200	Cultivated	Winter	7.59	56.91	11.88	81.48
3	1	c	200	Cultivated	Winter	8.32	62.38	9.08	116.67
3	1	a	300	Cultivated	Winter	8.10	60.77	16.28	65.74
3	1	b	300	Cultivated	Winter	7.97	59.81	6.68	104.63
3	1	c	300	Cultivated	Winter	6.60	49.51	78.28	88.89
3	2	a	100	Cultivated	Winter	8.15	61.09	44.28	85.19
3	2	b	100	Cultivated	Winter	7.12	53.37	49.08	77.78

3	2	c	100	Cultivated	Winter	6.13	45.97	43.88	44.44
3	2	a	300	Cultivated	Winter	4.07	30.52	35.88	88.89
3	2	b	300	Cultivated	Winter	3.34	25.04	19.48	78.70
3	2	c	300	Cultivated	Winter	3.34	25.04	20.68	43.52
3	3	a	300	Renosterveld	Winter	1.58	11.85	37.48	76.85
3	3	b	300	Renosterveld	Winter	0.64	4.76	41.88	73.15
3	3	c	300	Renosterveld	Winter	0.42	3.15	29.48	67.59
3	4	a	200	Renosterveld	Winter	0.08	0.58	20.68	98.15
3	4	b	200	Renosterveld	Winter	0.08	0.58	28.68	65.74
3	4	c	200	Renosterveld	Winter	0.00	0.00	5.08	62.96
1	1	a	200	Cultivated	Spring	9.97	74.76	43.08	21.30
1	1	b	200	Cultivated	Spring	4.77	35.75	10.28	65.74
1	1	c	200	Cultivated	Spring	7.23	54.25	22.68	57.41
1	1	a	300	Cultivated	Spring	3.11	23.31	19.88	52.78
1	1	b	300	Cultivated	Spring	2.48	18.60	26.68	13.89
1	1	c	300	Cultivated	Spring	5.57	41.80	2.28	62.04
1	2	a	150	Cultivated	Spring	2.93	21.96	14.68	49.07
1	2	b	150	Cultivated	Spring	3.74	28.02	5.88	50.93
1	2	c	150	Cultivated	Spring	5.17	38.78	3.48	100.00
1	2	a	300	Cultivated	Spring	0.96	7.16	10.28	103.70
1	2	b	300	Cultivated	Spring	1.49	11.20	16.68	28.70
1	2	c	300	Cultivated	Spring	1.22	9.18	11.48	50.93
1	3	a	150	Renosterveld	Spring	0.15	1.11	53.48	110.19
1	3	b	150	Renosterveld	Spring	0.00	0.00	3.48	65.74
1	3	c	150	Renosterveld	Spring	0.01	0.10	9.48	118.52
1	4	a	150	Renosterveld	Spring	0.00	0.00	37.48	94.44
1	4	b	150	Renosterveld	Spring	0.00	0.00	3.48	98.15
1	4	c	150	Renosterveld	Spring	0.00	0.00	0.00	74.07
2	1	a	200	Cultivated	Spring	6.56	49.20	54.68	129.63
2	1	b	200	Cultivated	Spring	7.50	56.27	23.88	65.74
2	1	c	200	Cultivated	Spring	6.78	50.89	9.08	64.81
2	2	a	200	Cultivated	Spring	5.80	43.49	42.68	109.26

2	2	b	200	Cultivated	Spring	4.59	34.41	44.68	60.19
2	2	c	200	Cultivated	Spring	4.63	34.74	3.08	109.26
2	3	a	150	Renosterveld	Spring	1.18	8.85	79.48	123.15
2	3	b	150	Renosterveld	Spring	0.51	3.80	41.48	106.48
2	3	c	150	Renosterveld	Spring	0.73	5.48	28.28	94.44
2	4	a	150	Renosterveld	Spring	0.64	4.81	43.08	56.48
2	4	b	150	Renosterveld	Spring	0.82	6.15	10.68	109.26
2	4	c	150	Renosterveld	Spring	0.46	3.46	57.08	74.07
3	1	a	200	Cultivated	Spring	9.48	71.07	29.88	155.56
3	1	b	200	Cultivated	Spring	8.22	61.65	5.88	154.63
3	1	c	200	Cultivated	Spring	7.73	57.95	6.68	98.15
3	1	a	300	Cultivated	Spring	7.23	54.25	8.68	104.63
3	1	b	300	Cultivated	Spring	7.05	52.90	25.88	90.74
3	1	c	300	Cultivated	Spring	6.78	50.89	82.28	109.26
3	2	a	100	Cultivated	Spring	7.77	58.28	62.28	172.22
3	2	b	100	Cultivated	Spring	9.34	70.06	16.68	167.59
3	2	c	100	Cultivated	Spring	9.12	68.37	16.68	143.52
3	2	a	300	Cultivated	Spring	3.11	23.31	13.48	71.30
3	2	b	300	Cultivated	Spring	3.02	22.63	5.48	96.30
3	2	c	300	Cultivated	Spring	3.83	28.69	3.08	53.70
3	3	a	300	Renosterveld	Spring	2.03	15.24	87.48	90.74
3	3	b	300	Renosterveld	Spring	2.30	17.25	41.48	75.00
3	3	c	300	Renosterveld	Spring	1.09	8.17	25.88	45.37
3	4	a	200	Renosterveld	Spring	0.73	5.48	35.88	63.89
3	4	b	200	Renosterveld	Spring	0.82	6.15	35.48	76.85
3	4	c	200	Renosterveld	Spring	0.64	4.81	37.88	38.89
1	1	a	200	Cultivated	Summer	4.39	30.71	173.08	48.15
1	1	b	200	Cultivated	Summer	4.07	28.50	91.48	58.33
1	1	c	200	Cultivated	Summer	4.21	29.45	85.08	39.81
1	1	a	300	Cultivated	Summer	2.47	17.31	83.08	50.93
1	1	b	300	Cultivated	Summer	2.50	17.47	40.68	51.85
1	1	c	300	Cultivated	Summer	2.72	19.05	57.08	63.89

1	2	a	150	Cultivated	Summer	3.87	27.09	123.88	60.19
1	2	b	150	Cultivated	Summer	2.38	16.68	113.88	68.52
1	2	c	150	Cultivated	Summer	2.79	19.52	88.28	80.56
1	2	a	300	Cultivated	Summer	2.61	18.26	192.28	30.56
1	2	b	300	Cultivated	Summer	2.07	14.47	187.48	103.70
1	2	c	300	Cultivated	Summer	1.89	13.21	180.28	83.33
1	3	a	150	Renosterveld	Summer	0.74	5.17	46.68	67.59
1	3	b	150	Renosterveld	Summer	0.56	3.91	62.68	57.41
1	3	c	150	Renosterveld	Summer	0.92	6.43	32.68	78.70
1	4	a	150	Renosterveld	Summer	0.58	4.07	111.08	34.26
1	4	b	150	Renosterveld	Summer	0.20	1.39	37.08	32.41
1	4	c	150	Renosterveld	Summer	0.00	0.00	94.68	20.37
2	1	a	200	Cultivated	Summer	8.40	58.77	44.28	40.74
2	1	b	200	Cultivated	Summer	8.87	62.09	91.48	61.11
2	1	c	200	Cultivated	Summer	8.35	58.46	33.48	81.48
2	2	a	200	Cultivated	Summer	7.14	49.95	86.68	73.15
2	2	b	200	Cultivated	Summer	6.75	47.27	110.28	75.93
2	2	c	200	Cultivated	Summer	6.46	45.22	126.28	57.41
2	3	a	150	Renosterveld	Summer	1.30	9.11	39.88	66.67
2	3	b	150	Renosterveld	Summer	1.41	9.90	34.28	60.19
2	3	c	150	Renosterveld	Summer	0.58	4.07	17.88	0.00
2	4	a	150	Renosterveld	Summer	1.08	7.54	27.88	52.78
2	4	b	150	Renosterveld	Summer	1.95	13.68	35.48	33.33
2	4	c	150	Renosterveld	Summer	0.74	5.17	37.08	62.04
3	1	a	200	Cultivated	Summer	15.42	107.96	15.88	58.33
3	1	b	200	Cultivated	Summer	12.09	84.63	17.88	44.44
3	1	c	200	Cultivated	Summer	10.40	72.81	13.08	52.78
3	1	a	300	Cultivated	Summer	11.19	78.32	20.68	63.89
3	1	b	300	Cultivated	Summer	12.02	84.16	24.68	53.70
3	1	c	300	Cultivated	Summer	9.79	68.55	20.68	55.56
3	2	a	100	Cultivated	Summer	13.96	97.72	18.28	61.11
3	2	b	100	Cultivated	Summer	10.63	74.38	15.48	43.52

3	2	c	100	Cultivated	Summer	11.01	77.06	15.88	50.00
3	2	a	300	Cultivated	Summer	8.42	58.93	18.68	25.93
3	2	b	300	Cultivated	Summer	6.82	47.74	37.88	51.85
3	2	c	300	Cultivated	Summer	6.73	47.11	14.68	39.81
3	3	a	300	Renosterveld	Summer	2.36	16.52	25.48	69.44
3	3	b	300	Renosterveld	Summer	1.41	9.90	24.68	32.41
3	3	c	300	Renosterveld	Summer	1.48	10.37	27.88	24.07
3	4	a	200	Renosterveld	Summer	1.86	13.05	21.88	19.44
3	4	b	200	Renosterveld	Summer	0.85	5.96	29.08	21.30
3	4	c	200	Renosterveld	Summer	1.12	7.85	24.68	45.37

Table C4 Phase 2 of soil analysis - Total P, N and C for sample sites in the study area as initially reported by Myburgh (2021) and reanalysed by the author at the BIOGRIP stable light isotope node at the University of Cape Town.

Transect	Site	Horizon	Date collected	Field position	Total P (mg/kg)	Total N (%)	Total C (%)
1	1	A	14/05/2021	Cultivated	436	0.18	1.86
1	2	A	14/05/2021	Cultivated	669	0.20	2.01
1	3	A	14/05/2021	Renosterveld	461	0.21	2.40
1	4	A	14/05/2021	Renosterveld	376	0.18	2.12
2	1	A	14/05/2021	Cultivated	357	0.12	1.27
2	2	A	14/05/2021	Cultivated	372	0.10	1.13
2	3	A	14/05/2021	Renosterveld	425	0.23	2.67
2	4	A	14/05/2021	Renosterveld	302	0.12	1.33
3	1	A	14/05/2021	Cultivated	535	0.16	1.86
3	2	A	14/05/2021	Cultivated	590	0.19	2.26
3	3	A	14/05/2021	Renosterveld	443	0.24	3.20
3	4	A	14/05/2021	Renosterveld	351	0.16	2.28
4	1	A	14/05/2021	Cultivated	522	0.15	1.56
4	2	A	14/05/2021	Cultivated	471	0.18	2.06
4	3	A	14/05/2021	Renosterveld	269	0.16	1.85
4	4	A	14/05/2021	Renosterveld	249	0.14	1.90
1	1	A	28/05/2021	Cultivated	611	0.24	2.79
1	2	A	28/05/2021	Cultivated	694	0.21	2.26
1	3	A	28/05/2021	Renosterveld	491	0.24	2.96
1	4	A	28/05/2021	Renosterveld	421	0.20	2.46
2	1	A	28/05/2021	Cultivated	404	0.15	1.58
2	2	A	28/05/2021	Cultivated	329	0.08	0.78
2	3	A	28/05/2021	Renosterveld	558	0.31	3.71
2	4	A	28/05/2021	Renosterveld	377	0.17	2.05
3	1	A	28/05/2021	Cultivated	646	0.17	1.85
3	2	A	28/05/2021	Cultivated	641	0.20	2.11
3	3	A	28/05/2021	Renosterveld	431	0.29	3.22
3	4	A	28/05/2021	Renosterveld	357	0.16	1.76
4	1	A	28/05/2021	Cultivated	540	0.17	1.77
4	2	A	28/05/2021	Cultivated	425	0.14	1.39
4	3	A	28/05/2021	Renosterveld	268	0.17	2.02
4	4	A	28/05/2021	Renosterveld	253	0.16	2.00
1	1	A	28/06/2021	Cultivated	604.8	0.21	1.81
1	2	A	28/06/2021	Cultivated	786.8	0.25	2.44
1	3	A	28/06/2021	Renosterveld	578.9	0.25	2.88
1	4	A	28/06/2021	Renosterveld	458.8	0.23	2.58
2	1	A	28/06/2021	Cultivated	496.6	0.18	1.88
2	2	A	28/06/2021	Cultivated	515.9	0.18	1.80
2	3	A	28/06/2021	Renosterveld	509.3	0.24	2.47
2	4	A	28/06/2021	Renosterveld	463.8	0.18	1.85
3	1	A	28/06/2021	Cultivated	841.6	0.22	2.35
3	2	A	28/06/2021	Cultivated	833.8	0.24	2.57
3	3	A	28/06/2021	Renosterveld	523.4	0.28	3.56
3	4	A	28/06/2021	Renosterveld	428.7	0.19	2.45
4	1	A	28/06/2021	Cultivated	638.5	0.16	1.63

4	2	A	28/06/2021	Cultivated	454.7	0.14	1.53
4	3	A	28/06/2021	Renosterveld	323.9	0.15	1.68
4	4	A	28/06/2021	Renosterveld	284.0	0.16	2.17

Table C5 N and C isotopic composition for soil samples collected during Phase 2.

			$\delta^{15}N$			$\delta^{13}C$		
Transect	**Site**	**Horizon**	**14-May**	**28-May**	**28-Jun**	**14-May**	**28-May**	**28-Jun**
1	1	A	4.04	3.68	4.16	-26.58	-27	-26.64
1	2	A	3.90	3.88	3.80	-26.67	-27	-26.91
1	3	A	4.41	4.24	4.23	-25.71	-26.42	-26.42
1	4	A	4.42	4.13	4.33	-26.23	-26.35	-26.37
2	1	A	4.13	3.79	3.51	-26.83	-27.26	-27.93
2	2	A	4.46	4.72	3.99	-26.65	-26.27	-28.11
2	3	A	4.90	4.67	4.94	-27.44	-27.65	-27.93
2	4	A	4.79	4.95	5.20	-27.28	-27.74	-28.43
3	1	A	4.47	4.71	4.32	-27.04	-27.2365	-27.04
3	2	A	4.28	4.43	4.34	-27.13	-27.1405	-27.74
3	3	A	4.78	5.04	4.83	-27.26	-27.293	-27.91
3	4	A	4.98	5.18	4.73	-26.57	-26.976	-27.92
4	1	A	4.37	3.99	4.05	-26.68	-27.1523	-27.20
4	2	A	3.78	4.42	4.26	-27.32	-26.7908	-27.29
4	3	A	5.10	4.93	5.36	-26.58	-27.1355	-26.96
4	4	A	5.08	5.20	5.09	-26.06	-26.4005	-26.31
Fertiliser A			-1.55			-42.77		
Fertiliser B			-1.53			-42.61		
Fertiliser C			-1.50			-42.77		