Changes in the Berg River Basin over time

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Summary

The nature of river ecosystems is influenced by the history of activities in their basins. This dissertation investigated historic changes in the Berg River Basin and their influence on river ecosystem structure. The central assumption was that all activities in a river basin landscape contribute either directly or indirectly to the condition (physical, biological, chemical) of rivers that run through them. It was first necessary to establish what changes had taken place in the river basin over time and this was done in different ways at different spatial scales. Changes in land-use were collated and mapped across the basin since these were considered to influence the river's flow regime and river channel structure. Predictions were made about how changes in flow and river channel habitat would influence the distribution and abundance of aquatic macroinvertebrates.

A history of land-use changes over the Berg River Basin was explored between four periods, 1955-1965, 1976-1985, 1996-2005 and 2006-2015. The bulk of the dryland crop production was in the lower foothills and lowlands while the upper foothills comprised orchards, vineyards and forestry. From 1955-2015 the extent of agricultural land in the basin declined by half as dryland crops were changed to orchards and vineyards and large tracts of land were left fallow. Over the same period the area under forest declined by 73% and urban areas doubled in size as did the number of farm dams in response to the increased need for irrigation to supply the more water hungry crops. The effects of the changes in land-use, the increase in farm dams and the construction of large dams on the river's flow regime was investigated next.

Changes in flow were explored at four river gauges along the length of the Berg River up- and down-stream of the two main in channel dams; the Berg River Dam in the Upper Foothills and Misverstand Dam in the Lowlands. In general the changes were more marked at the downstream gauges and the trends were towards increased dry season flows and slightly decreased wet season flows due to release of water from, and capturing of floods by the in-channel dams to meet irrigation demand in the dry season. Flow pattern from early records was better correlated with rainfall than that from the recent record indicating that flow changes were likely to be attributable to anthropogenic effects such as land-use and water resource developments. Both land-use and water resource developments were predicted to have consequences on river channel shape and habitat that was investigated next.

Changes in river channel shape, the extent and composition of the floodplain and riparian area was mapped from aerial photographs at five sites along the Berg River and at five adjacent tributaries. Each site responded differently, which was not unexpected, and reductions in the extent of the channel and riparian area were more severe along the Berg River main stem when compared to the tributaries. Along tributaries no floodplains were discernible at the scale measured, however a decreased in extent over time along the main river except downstream of the Berg River Dam where the floodplain area had increased due to the previously braided channel of 1938 changing to a single thread channel with floodplain and a greater area of sandbanks. Changes in river habitat, such as these, were predicted to

effect change in the abundance and community structure of aquatic macroinvertebrates, which was investigated next.

The abundance of aquatic macroinvertebrates was studied from the 1950s to 2015 and showed a reduction in simulids and baetids with an increase in the abundance of chironomids, indicating a decline in water quality. Changes in other groups indicated a decline in quality of habitat, for instance a loss of plecopterans that prefer clean gravel beds being replaced by caenids that prefer a sandy channel bottom. In 2015 there were also more groups of invertebrates that are associated with slow-flowing areas and marginal vegetation, which was presumed to have occurred in response to the clearing of woody alien trees from the river banks and the subsequent proliferation of aquatic and marginal plants along the water's edge.

Data collected for land-use, hydrology, channel and riparian changes, macroinvertebrates were synthesized using BEST (BIOENV and BVSTEP) multivariate statistics in PRIMER to search for high rank correlations between environmental and biological variables. When the environmental variables were tested against the biological variables showed that changes in macroinvertebrates were strongly related to area of plantations, area of undeveloped land, the extent of braiding, maximum 5-day average discharge in the wet season and the daily average volume in the dry season. Environmental variables were most influenced/driven by location (separated into sub-basins) while time was the driving factor for the macroinvertebrates data.

Opsomming

Die aard van rivier-ekostelsels word deur die geskiedenis van hul aktiwiteite in hul komme beïnvloed. Hierdie studie was ŉ ondersoek na historiese veranderinge in die Bergrivierkom en die invloed daarvan op rivier-ekostelselstruktuur. Die sentrale aanname was dat alle aktiwiteite in die rivierkomlandskap direk of indirek tot die toestand (fisies, biologies, chemies) van riviere wat daardeur vloei, bydra. Eerstens moet bepaal word watter veranderinge in die rivierkom met verloop van tyd plaasgevind het, wat op verskillende maniere teen verskillende ruimteskale uitgevoer is. Veranderinge in grondgebruik is noukeurig vergelyk en oor die kom gekarteer, aangesien dit beskou is as verantwoordelik vir die riviervloeistelsel en rivierkanaalstruktuur. Voorspellings is gemaak oor hoe veranderinge in vloei en rivierkanaalhabitat die verspreiding en oorvloed van watermakro-invertebrate sou beïnvloed.

ŉ Geskiedenis van grondgebruikveranderinge in die Bergrivierkom in vier tydperke, naamlik 1955–1965, 1976–1985, 1996–2005 en 2006–2015, is ondersoek. Die meerderheid droëlandgewasproduksie het in die laer voetheuwels en laaglande plaasgevind, terwyl die boonste voetheuwels uit boorde, wingerde en boswêreld bestaan het. Van 1955 tot 2015 het die omvang van landbougrond in die kom met die helfte afgeneem, aangesien droëlandgewasse na boorde en wingerde verander is en groot landstreke braak gelaat is. In dieselfde tydperk het die bosgebied met 73% afgeneem en stedelike gebiede het in grootte verdubbel, en so ook die aantal plaasdamme in reaksie op die verhoogde vraag na besproeiing om aan die waterhonger gewasse te verskaf. Die gevolge van die veranderinge in grondgebruik, die toename in plaasdamme en die bou van groot damme in die rivier se vloeistelsel is hierna ondersoek.

Veranderinge in vloei is by vier riviermeters teen die lengte van die Bergrivier hoër op en laer af van die twee hoofdamme in die kanaal, die Bergrivierdam in die boonste voetheuwels en Misverstanddam in die laaglande, ondersoek. In die algemeen was die veranderinge meer opvallend by die stroomafmeters en die neigings was na verhoogde droëseisoenvloei en effens verlaagde natseisoenvloei weens die vrystelling van water vanaf en opvangs van strome deur die damme in die kanaal om in besproeiingsbehoeftes in die droë seisoen te voorsien. Die vloeipatroon van vroeë rekords is noukeuriger met reënval as in die vorige rekord vergelyk, en het getoon dat vloeiveranderinge waarskynlik aan antropogeniese gevolge soos grondgebruik en waterhulpbronontwikkeling toegeskryf kan word. Daar is voorspel dat sowel grondgebruik as waterhulpbronontwikkeling gevolge vir rivierkanaalvorm en habitat inhou, wat vervolgens ondersoek is.

Veranderinge in rivierkanaalvorm en die omvang en samestelling van die vloedvlakte en oewergebied is van lugfoto's by vyf terreine langs die Bergrivier en vyf aangrensende takriviere gekarteer. Al die terreine het verskillend gereageer, wat nie onverwags was nie, en verlaging van die omvang van die kanaal en oewergebied was groter langs die Bergrivier-hoofrivier in vergelyking met die takriviere. Geen vloedvlaktes is langs die takriviere teen die gemete skaal waargeneem nie, alhoewel ŉ afname in omvang met verloop van tyd langs die hoofrivier af waargeneem is, maar wel nie laer af van die Bergrivierdam nie, waar die

vloedvlakte-oppervlakte toegeneem het weens die vorige vlegkanaal van 1938, wat in ŉ enkeldraadkanaal met vloedvlakte en ŉ groter oppervlakte sandbanke verander het. Die voorspelling is gemaak dat veranderinge in rivierhabitat, soos hierdie, verandering in die oorvloed en gemeenskapstruktuur van watermakro-invertebrate teweeg sou bring, wat vervolgens ondersoek is.

Die oorvloed watermakro-invertebrate vanaf die 1950's tot 2015 is bestudeer en het ŉ afname in simuliede en baetiede getoon, met ŉ toename in die oorvloed chironomiede, wat op ŉ afname in watergehalte dui. Veranderinge in ander groepe dui op ŉ afname in habitatgehalte, byvoorbeeld ŉ verlies aan plekopterane wat skoon gruisbeddings verkies, wat met kaeniede vervang is, wat ŉ sandkanaalbodem verkies. In 2015 was daar ook meer groepe invertebrate wat met stadig vloeiende gebiede en randplantegroei geassosieer word, en die aanname is gemaak dat dit plaasgevind het in reaksie op die uitwissing van houtagtige uitheemse bome aan die rivieroewers en die gevolglike voortplanting van water- en randplante teen die waterrand.

Data wat vir grondgebruik, hidrologie, kanaal- en oewerveranderinge en makro-invertebrate ingesamel is, is met behulp van BEST (BIOENV and BVSTEP) meerveranderlike statistieke in PRIMER gesintetiseer om hooggeklassifiseerde verbande tussen omgewings- en biologiese veranderlikes te vind. Met toetsing van die omgewingsveranderlikes teen die biologiese veranderlikes, is gevind dat veranderinge in makro-invertebrate sterk verband hou met die oppervlakte van plantasies, die oppervlakte van onontwikkelde grond, die omvang van omvlegting, maksimum vyfdag- gemiddelde afloop in die nat seisoen en die daaglikse gemiddelde volume in die droë seisoen. Omgewingsveranderlikes is die meeste deur ligging (in subkomme geskei) beïnvloed/aangedryf, terwyl tyd die dryffaktor vir die data oor makro-invertebrate was.

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1 Introduction

1.1 Background

River systems run through different landscapes that change over time (Dorava *et al.* 2001). Changes in landscapes may be caused by natural/environmental or human beings through modifications of the land-use and land cover. This landscape influences and determines the river's ecosystems and its overall ecological condition (Naiman *et al.* 2005). This dissertation seeks to quantify the links between a river system and the landscapes it drains, by explaining how the physical drivers of river condition (land-use, flow and riparian vegetation) influence the responses of aquatic biota (macroinvertebrates). The need to understand this links is important in order to understand and predict how aquatic biotas (macroinvertebrate) respond to change in surrounding land-use and flow characteristics. This is motivated by the central assumption that all activities that happen within a river basin contribute to the river's ecological state (Naiman *et al.* 2005). The intention is that if possible at the end of this project, the large and small scale information that has been acquired from all chapters can be integrated to furthermore develop a protocol that can be replicated in other river basins through establishing a framework that may be used for monitoring and updating the Ecological Reserve after implementation.

In the past, water resource management in South Africa was largely supply driven and non-ecological in nature (Roux 1999; Stein 2005; Nomquphu *et al.* 2007). Initially the allocation of water for the environment was based on the need to reserve minimum flows for survival of specific species of importance (Mazvimavi *et al.* 2007). In many parts of the world this is still the case, while in others there is little scientific input concerning the water needs of freshwater ecosystems (Richter *et al.* 2006). In South Africa, however, the National Water Act 36 of 1998 (NWA 1998) made provision for the 'Reserve', which is defined as: 1) the quantity and quality of water required to satisfy Basic Human Needs (BHN) by securing a basic water supply as prescribed under the Water Services Act 1997 (Act no. 108 of 1997) for people who rely on water from the relevant water resources, and; 2) the quantity, quality and distribution in time of water to protect aquatic ecosystems so as to ensure ecologically-sustainable development and use of the relevant water resources. The latter is referred to as the "Ecological Reserve".

An Ecological Reserve refers to the quantity and quality of water that is set aside in order to provide for the human basic need and an allocation of water for the aquatic environment (NWA 1998). Data on flow, water quality, aquatic macroinvertebrates, riparian vegetation and other indicators are routinely collected at monitoring sites by Department of Water and Sanitation (DWS) personnel (e.g., River Eco-Status Monitoring Programme (REMP) previously known as the River Health Programme (RHP)). These data are used in a number of management processes, including monitoring the ecological condition of rivers and direct implementation of the Ecological Reserve.

Determination of the Ecological Reserve and other decisions on river basin management are based on a combination of biophysical data that are used in an integrated approach that assesses the physical river conditions in an ecologically-relevant way. These include consideration of data sets of flow, water quality, aquatic macroinvertebrates, fish, riparian vegetation, habitat and other biophysical parameters that are collected at sites along rivers (Roux 1999; RHP 2004). However, there is a need to reconcile site-specific data in a meaningful way with large-scale processes. Furthermore, broad consideration of issues that influence river ecosystem health are not a routine part of river monitoring and water-recource management (McDonnell 2008). Since the nature of lotic ecosystems within a river basin is largely defined and controlled by the landscapes that surround them, and river health is, at least in part, driven by large-scale processes of water and sediment transported across the lotic landscape (Petts 1994; Dorava *et al.* 2001; Ward *et al.* 2001; Naiman *et al.* 2005), the landscape approach provides vital context for the evaluation of changes in a river.

The interaction between land, river and elements of the hydrological cycle are illustrated in the lotic landscape (Figure 1.1) and comprise four main habitats; land, the riparian area and floodplain, the river banks and channel, and the hyporheos.

Climate, topography, geology and land-use influence the hydrology of the basin (flow and rainfall), structure and pattern of the channel and its riparia and ecosystem functioning (both aquatic and terrestrial).

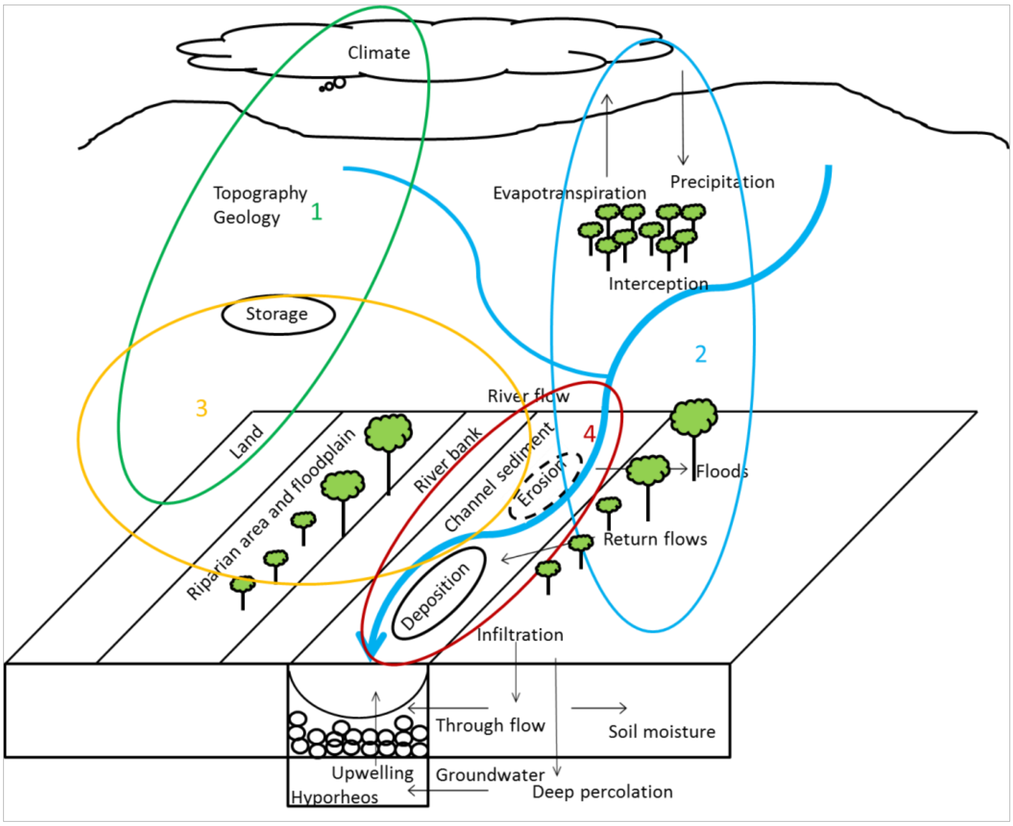

Figure 1.1 Conceptual framework showing processes of the river basin and the four focus areas of the PhD. 1 = large scale land-use changes; 2 = hydrological characteristics; 3 = riparian and channel planform of the river and 4 = aquatic biota (macroinvertebrates) of the basin

River ecosystems are defined by, and profoundly affected by changes to, processes within their basins (Dorava *et al.* 2001; Naiman *et al.* 2005). The central hypothesis of this dissertation is that all activities in the lotic landscape contribute either directly or indirectly to a river's ecological condition, as defined broadly by physical, biological and chemical attributes (Harald *et al.* 1975; Dorava *et al.* 2001; Naiman *et al.* 2005). The cumulative result of the many impacts may lead to changes in basic structure and function of the riverine-riparian ecosystems resulting in a reduced ability to perform ecological functions (Dopplet *et al.* 2012). With this in mind, understanding the natural and historic fluctuations within river ecosystems is essential to provide a context within which data that are generated by monitoring programmes could be assessed. There is a growing awareness to incorporate these large-scale and less-obvious causes of change into judgements made with respect to river condition (Opperman and Harrison 2008; Doppelt *et al.* 2012). This study enhances our current understanding of the natural and historic fluctuations within river ecosystems, and their drivers, at a river basin scale through the use of long-term data sets of historic impacts and ecosystem condition. The main aim of the study is to document, analyse and assess large-scale biophysical data that are readily available to help inform monitoring of rivers. The four ovals (Figure 1.1) highlight the four focus areas of the PhD dissertation using the linked elements of the hydrological system on the lotic landscape.

The study was based on the framework that the river basin landscape provides context for the river determined by its geological and climatic properties as well as the historical changes that have taken place. The schematic diagram uses the hydrological cycle to illustrate the river structure and function within the lotic landscape (Figure 1.1). During precipitation water either; infiltrates the soil, is intercepted by vegetation, runs off over land or immediately evaporates (Figure 1.1 - 2: blue oval). The path taken affects the length of time that a water drop spends in the basin and leaches nutrients and minerals from the surrounding rocks and soils, which determine its chemical nature. The length of the journey, and the quality of the water at the end of the journey, are affected by four main, interlinked, variables; climate, geology, topography and the nature and extent of the vegetation cover as well as the human-induced changes (1: green oval). Floods influence the nature and structure of the riparian area and flood plain (see Section 2.1), which in turn influences the structure and character of the bank and bed habitats, and the flow between these (Figure 1.1 - 3: orange oval).

In accordance with the focus areas, five primary objectives of the study were as follows:
(1) To document large-scale land-use changes in the lotic landscape over space and time (green oval); this objective was addressed using the green oval by using a broader view that considers the interactions between the river and its surrounding land as water, organic matter, sediments, nutrients and biota move through the river basin. The length of the journey and the quality of the water at the end of the journey, are affected by four main, interlinked, variables; climate, geology, topography, land-use and the nature and extent of the vegetation cover.
(2) To examine the temporal hydrological characteristics of the river basin by analysing the historical hydrology of the Berg River and identify major changes in the volume or seasonal distribution of flow, and where possible identify the underlying causes thereof (blue oval).
(3) To identify changes in river channel structure and the riparian area over time and where possible to link these with changes in land-use and flow along the river. The combined

action of all activities that are happening at all parts of the basin are also reflected on the changes of the properties of the channel and its riparian area over time (Figure 1.1 - orange oval).

(4) To identify patterns in aquatic macroinvertebrate communities in the Berg River over time and to compare these to other changes in the basin and river in an effort to identify key drivers of change. The combined action of all activities that are happening at all parts of the basin are also reflected on the changes of the properties of the aquatic biota over time (Figure 1.1 - red oval).

(5) A fifth and overarching objective was to use the insights gained to develop a framework for the collation and evaluation of similar data, as and when available, in other basins with a view to improving the spatial and temporal context within which ecological monitoring data are evaluated. The approach was to build a database of different information layers, such as topography, land-use type, urban development and agricultural areas, hydrology and macroinvertebrates, each representing particular periods in the history of the basin from as early as possible with various time-layers from *c*. 1900 to 2015.

1.2 Definitions

The following definitions have been applied in this dissertation:

River basin	The area of land drained by a river and its tributaries. This can refer to a first order river, a tributary of a larger river or a whole river system (Frissell *et al.* 1986; Allan *et al.* 1997).
Lotic landscape	The landscape in which lotic aquatic ecosystems occur, define and control almost every facet of their nature and functioning. For this reason, we have referred to the river basin as the 'lotic landscape', defined as comprising all parts of the basin - the terrestrial portions plus the lentic and lotic systems that drain them. The lotic landscape comprises the landscape across which water from rain and/or melting snow/ice drains to a single point at a lower elevation, from whence they join another waterbody, such as a lake or the sea (Davies and Day 1998; Finlayson and McMahon 2004).
Land-use	Purpose to which the land cover is committed to, encompasses all kinds of human uses of the basins, including vegetation/crop type, roads, buildings and other catchment hardening, infilling of wetlands, farm dams, major water-resource infrastructure and impoundments, inter-basin transfers, diversions of rivers; it excludes revetments, channelization or canalisation (Clawson 1965; Australian Government Bureau of Rural Sciences, 2006).
Environmental Flows	Water that is left in a river system, or released into it, for the specific purpose of managing the ecological condition of that river (King *et al.* 2003a).
Ecological Reserve	Refers to the volume and quality of flow that is reserved for maintenance of the aquatic ecosystems of the water resource (Yang *et al.* 2009), this also vary depending on the class of the resource (Belcher 2004).

Resource Quality Objectives	Are quantitative and qualitative descriptions of the hydrological, chemical, physico-chemical, geomorphological and biological attributes that can be monitored for compliance of the management classes that have been set (Ashton 2012). This includes the volume and timing of flows that are required for the Ecological Reserve (King and Pienaar 2011).
Thalweg	The line of the deepest or lowest elevation within the channel which can also be the middle water course (Bakhashab 1996)
Riparian zone	Refers to areas directly adjacent to the active channel of a water course or waterbody that support vegetation communities which are distinctly different to neighbouring terrestrial communities (Naiman *et al.* 2005; Reinecke *et al.* 2007)
Intra-annual flood	Floods with a return period of less than one year
Inter-annual flood	Floods with a return period of greater than one year
Aquatic macroinvertebrates	Animals without back bone that which live a part of their lives in freshwater biotopes (Machay & Eastburn 1990).

1.3 Chapter overview

The PhD dissertation comprises five data chapters (Chapters 3-7), plus two introductory chapters (Chapters 1 and 2) and a concluding chapter (Chapter 8), as follows:

- Chapter 1: Introduction and motivation for the study
- Chapter 2: Literature Review
- Chapter 3: Historical changes in land-use of the Berg River Basin. This examines large-scale changes in the lotic landscape of the Berg River.
- Chapter 4: Historical changes in the flow regime of the Berg River. This looks at historical hydrology and identifies the causes of major disruptions in either the volume or distribution of flows in the Berg River.
- Chapter 5: Historical changes in channel planform of the Berg River. This examines changes to the riparian area and/or floodplain, river bank and river channel of the Berg River using historical data to analyse changes over time.
- Chapter 6: Historical changes in aquatic macroinvertebrate communities in the Berg River. This examines changes in the macroinvertebrate communities of the Berg River using historical data to analyse changes over time. The analysis will interpret these changes in terms of the habitat changes described in Chapter 5.
- Chapter 7: A proposed framework for the use of historic data to support the River Eco-Status Monitoring Programme. This last chapter syntheses the four data chapters in order to find correlations between the available environmental data (land-use, hydrology, channel and riparian change) and the biological data (macroinvertebrates).
- Chapter 8: Conclusion

There is also a reference list (Chapter 9) and a series of appendices (Appendix A to Appendix C) that contain additional information of relevance to various data chapters. This study was part of a much bigger Water Research Commission "*A framework for using historic information to aid monitoring the ecological Reserve*", WRC project No. K5-2345.

2 Literature Review

2.1 Lotic systems

The landscapes in which lotic ecosystems develop, define and control almost every facet of their nature and functioning (Dorava *et al.* 2001; Naiman *et al.* 2005). For this reason, the basin has been referred to as the 'lotic landscape', defined as the terrestrial portions of the river basin, plus the lentic and lotic systems that drain them. The lotic landscape comprises the landscape across which water from rain and/or melting snow/ice drains to a lower elevation, ultimately joining another waterbody, such as a lake or the sea (Davies and Day 1998; Finlayson and McMahon 2004). This landscape provides a context for the river, determined by its geological properties, the prevailing climate and the historical changes that have taken place.

Two broad types of aquatic ecosystems can occur in basins: lotic ecosystems, which are dominated by the unidirectional flow of water, and include springs, streams, rivers and portions of wetlands (Fisher *et al.* 2007), and; lentic ecosystems, where the water does not have unidirectional flow, such as ponds, marshes and lakes (Chapman and Reiss 1998). They are separate but connected parts of the lotic landscape, through which water, sediments, nutrients and other elements flow, and both are an important part of, and interact closely with, the surrounding lotic landscape (Ward 1989).

The number and size of streams forming the drainage network through this landscape is determined largely by a combination of area, topography, climate and geology (Finlayson and McMahon 2004). Geomorphic and geologic characteristics of basins such as substrates, faulting, rock types and their mineral content affect the quality of the water draining them, as well as the ground-surface water interactions within them (Leopold 1994; Finlayson and McMahon 2004; Bell 2007). Together these characteristics govern the vegetation of their basins (Bendix and Hupp 2000), the spatial and temporal distribution of the water and sediments supplied to a river, and the river's ability to arrange and transport these, thereby creating the physical habitat on which the biota exist (Poff *et al.* 1997; Burgmer *et al.* 2006). These multiple influences operate at different spatial scales (Allan and Johnson 1997), with each pair of points in the drainage network connected by a unique one-dimensional path (Rodriguez-Iturbe and Rinaldo 2001). In the rivers, different morphological units, such as mid-channel bars, terraces and lateral bars, support different biotic communities either due to the processes that are active on them (principally flooding) or because of their physical characteristics such as substrate type (Bendix and Hupp 2000). Substrate size and type influences such diverse aspects of the system as porosity, which affects surface-groundwater interchange, availability of nutrients (Sher and Marshall 2003), suitability for penetration by plants roots, and suitability of habitat for spawning and/or refuge. For instance, rivers with unstable substrates usually have low species diversity (Bunn and Arthington 2002) as they are perilous for the small aquatic life that form the basis of the food chain.

Lotic systems interact with their surrounding basins over four dimensions (Amoros *et al.* 1987 in Finlayson and McMahon 2004; Ward and Tockner 2001):

- longitudinally - upstream to downstream progression;
- laterally - across the main channel and riparian area including floodplains;
- vertically - through the hypoheric zone;
- temporally - through time: daily, seasonal and annual changes in river dynamics and ecosystem functioning.

Rivers usually rise in the mountains, either as a springs or seeps and begin their journey to the sea by flowing down steep, narrow channels, with clear fast-flowing water (Davies and Day 1998). Here the beds tend to be coarse with large boulders and rocky outcrops (Finlayson and McMahon 2004; Davies and Day 1998). These upper reaches are the production zone in that they usually generate the bulk of the water and sediments that are transported downstream (Rodriguez-Iturbe and Rinaldo 2001; Finlayson and McMahon 2004). Out of the mountains, in the middle or foothill reaches (Rowntree *et al.* 2000), the slope tends to be gentler, and the river bed widens, reducing the flow velocity (Davies and Day 1998). Additional flow is delivered mainly through tributaries. These middle reaches are the transport or transfer zone where both erosion and deposition both take place (Rodriguez-Iturbe and Rinaldo 2001). The water is less pure and more turbid than in the upper reaches (Davies and Day 1998). Out of the foothills, in the lower reaches, the river valley is wider allowing more lateral movement and meandering (Allan and Castillo, 2007), especially on the coastal plain. These lower reaches are predominately the deposition zones (Finlayson and McMahon 2004). Discharge is highest of all the reaches, as increasingly more tributaries join the main river (e.g., Allan and Castillo 2007), but the river slope is flatter and flow velocity drops resulting in sediment deposition (Rodriguez-Iturbe and Rinaldo 2001). The overall the characteristics of a river system in any one of these reaches, however, depends on the transfer of water, sediments, nutrients and organic matter (Petts and Foster 1985), and the interaction and association of river biota is in relation to the nature and conditions imposed by these (Finlayson and McMahon2004). Longitudinal connectivity from upstream to downstream is provided by the channels. For a river channel to form it requires sufficient water discharge and slope to erode and transport surface material (Bridge 2009). Typically, the channel is distinguishable from the surrounding areas by a marked increase in water velocity and, in the lower reaches, by natural leeves (Junk *et al.* 1989). During the initial stages of development, channels tend to accommodate to local geography and geological structures, developing along fault zones (Finlayson and McMahon 2004; Bell 2007). However, as water, sediments, and organic debris are routed through the drainage network, channel form is shaped to an ever greater degree by processes that govern the supply and transport of these materials (Montgomery 1999; Kleinhans 2010). For instance, an increase in sediment load with less transport capacity (less water) may result in braided channels whereas a reduction in sediment load may result in meandering of a channel (Church 2006).

The width of the channel is also determined by the processes of floodplain formation and destruction (Kleinhans 2010). River channels and their associated riparian areas increase in width and complexity with longitudinal distance from the source as the balance between sediment supply and transport shifts from supply limited channels upstream to transport limited channels downstream. The study of river morphology groups parts of the drainage network according their relationship to their source (Rodriguez-Iturbe and Rinaldo 2001); channels that originate at the source (spring or seep) with no tributaries are called first order streams; when two first order streams join they form a second order stream and as more streams join the system network the order increases; assuming that when two channels of order N join, they form a channel of order $N + 1$ (Stahler 1957; Scheidegger 1965). As tributaries join each other, channel networks arrange in different drainage patterns such as dendritic, trellis, rectangular, radial, angular and parallel (Zernitz 1932). Generally the size of river's drainage network increases downstream as tributaries and groundwater contribute to flow (Allan and Castillo, 2007). As is the case with the main channel, tributaries have unique characteristics based on the landscapes they drain and the in-channel processes these support (Finlayson and McMahon 2004).

Riparian areas embody the lateral interface between the river channel and the surrounding landscape and comprise a diverse mosaic of landforms, communities and environments within the larger landscape (Naiman *et al.* 1993; Naiman and Decamps 1997). These areas are periodically affected by flow and material transfer and, ecologically, are a transition zone between aquatic and terrestrial ecosystems (Jackson and Fisher 1986). Vegetation in these corridors grows in distinct lateral bands, broadly defined by the frequency with which they are inundated (Reinecke and Brown 2013). Where the gradient is gentle, riparian corridors may take the form of floodplains, which are flooded regularly during the wet season by large events that spread out over the area, and then drain back into the channel slowly, often only during the following dry season. As such, floodplain communities are also adapted to seasonal changes in inundation, nutrients, and light (Junk *et al.* 1989). Since the water on floodplains is usually slow flowing and/or still, they behave more like lentic than lotic systems (Davies and Day 1998). These are highly productive systems[1], as nutrients and sediments from the surrounding basin are carried onto the floodplain by the strongly flowing floodwaters and then left behind by the slow returning flows.

Riparian corridors and floodplains are among the most structurally-complex and biologically-diverse terrestrial landscapes on earth (Lorenz *et al.* 1997; Ward *et al.* 2001; Merritt and Wohl 2002). These are the areas directly adjacent to the wetted channel of a river that support vegetation communities, which are distinctly different from neighbouring terrestrial communities (Reinecke *et al.* 2007), where the vegetation typically shows a distributional relationship to the flow regime of the river (Reinecke and Brown 2013). The life cycles of many riparian species have been found to be closely linked to the natural variation of flow of a river (Poff *et al.* 1997; Friedman and Auble 2000; Pettit *et al.* 2001). The vegetation zones themselves occupy a three-dimensional transitional area (longitudinal, lateral and vertical) between aquatic and terrestrial ecosystems and serve as a passageway for the exchange of materials and energy from one ecosystem to the other (Naiman and Decamps 1997; Naiman *et al.* 2005; Kondolf *et al.* 2006; Reinecke *et al.* 2007; Richardson *et al.* 2007). The erosional and depositional forces of floods create different aquatic habitats that are suitable for refuge, spawning and feeding (Aarts *et al.* 2002), floods also exchange materials (nutrients and sediments) laterally between the channel and the floodplain.

Healthy riparian areas help to maintain the form of rivers by binding soils and strengthening river banks (Thorne 1990). Trees and shrubs increase channel roughness, thus resistance

[1] As such they are often targeted for agriculture (Bayley 1995, Downs and Gregory 2014).

to flow, which reduces the velocity of the flow in the channel and may lead to deposition of fine sediments and seeds in these areas (Chaimson 1989; King *et al.* 2003a). Riparian vegetation also acts as a buffer against sediments, fertilizers, pesticides and other matter draining from the surrounding landscape through direct chemical uptake (Lorenz *et al.* 1997; Dosskey *et al.* 2010). Riparian areas and floodplains also provide migratory corridors for animals and breeding; feeding or nursery grounds for a variety of floral/faunal communities (Brode and Bury 1984; Naiman *et al.* 1993; Corbacho *et al.* 2003), and provide food and shelter for people and wildlife.

The vertical interface, between surface and groundwater environments, occurs beneath and alongside a stream bed in an area known as the hyporheos (Vallet *et al.* 1993; White 1993). The hyporheic zone is an integral component of rivers, which expands the spatial extent of lotic ecosystem (Ward 1989), and provides critical habitat and refuge for many riverine organisms (Vallet *et al.* 1993). Here subsurface flows provide nutrients to the river and surface flows provide dissolved oxygen and organic matter to microbes and invertebrates underground (Stanford and Ward 1993). This zone varies in space and time making it difficult to identify, but the processes and pattern of underflow and discharge depend on proximity of water table to the surface, channel bed permeability, streamflow level, bedrock geology and topography (Boulton *et al.* 1998; Wiens 2002; Finlayson and McMahon 2004). In a river basin, the hyporheos tends to be most significant in the middle reaches (Stanford and Ward 1993; Boulton *et al.* 1998), although floodplains of large alluvial rivers are also characterized by high volumes of hyporheic flow (Stanford and Ward 1993). Although not much is known about the foodweb dynamics in the hyporheos (Stanford and Ward 1993), the ecosystem consists of interacting physical, chemical, and biological processes (Hakenkamp *et al.* 1993; Hendricks 1993; Valett *et al.* 1993). Different communities of organisms are said to be interacting within this ecotone, such as insects with hypogean (underground) and epigean (surface) life history stages as well as obligate groundwater species (Stanford and Ward 1993; White 1993). Depending on the bedrock geomorphology water from an unconfined aquifer upwells directly into the channel or floodplain (Stanford and Ward 1993).

The spatial and temporal variability inherent in the links between hydrologic, geomorphic and ecological processes are the fourth dimension affecting the nature and functioning of river ecosystems (Montgomery 1999). Flowing water erodes bedrock and soils, and redistributes alluvium (Standford 1998), thus the pattern and variety of flows ultimately determines both the landscape and the nature of the lotic systems draining it as well as the biota it hosts (Poff *et al.* 1997). Natural variations in fluvial action (erosion, sediment transport, deposition) create and maintain a high diversity of morphological units, such as pools, riffles, runs, gravel bars, avulsion channels, islands, debris dams and lateral floodplain terraces (Stanford *et al.* 1996), making the land-water interface both complex and dynamic (Allan 2004).

The flow regime also exerts direct control over the abundance and spatial arrangement of individuals, their life-history traits and their response to adverse conditions (Poff and Ward 1989; Bunn and Arthington 2002). The flow of water transported sediments through the channel; from uplands through the middle reaches to the lowlands of the basin, the higher uplands are erosion dominated while the lowlands are dominated by deposition (Rodriguez-Iturbe and Rinaldo 2001). Disturbance (floods and intermittency) and flow variability act on the physical template (Poff and Ward 1989). Flows create and destroy habitats in the channel, in the riparian corridor and on the floodplains (Naiman *et al.* 1993; Aarts *et al.* 2002). Flow, in particular floods, also transports plant propagules and nutrients (Friedman and Auble 2000), and provides the medium in which aquatic life can flourish. The flow regime of a river is comprised of different kinds of flow (low and high flow; small, large and larger floods), each of which contributes to the river's character and maintenance (Dunne and Leopold 1978; Booth and Jackson 1997; Paul and Meyer 2001; Brown and King 2002; Allan 2004):

- Low-flows are the flows in the river outside of floods. They maintain the basic ephemeral, seasonal or perennial nature of the river, thereby determining which animals and plants can survive there. The different magnitudes of low-flow in the dry and wet seasons create more or less wetted habitat and different hydraulic and chemical conditions, which directly influence the balance of species. For instance, species which need to spend several months in water to complete their life-cycles are rare in temporary rivers, though specific riparian tree species may be able to live on such a river's banks if the groundwater conditions are favourable.
- Small floods occur several times within a year. They stimulate spawning in fish, flush out poor quality water, mobilise sandy sediments, and contribute to flow variability. They re-set a wide spectrum of conditions in the river, triggering and synchronising activities as varied as upstream migration of fish and germination of riparian seedlings.
- Large floods occur more rarely than once a year. They dictate the general geomorphological character, shape and size of a river channel. Floods mobilise sediments and deposit silt, nutrients and seeds on floodplains. They inundate backwater areas, and trigger the emergence of adults of aquatic insects, which provide food for fish, frogs and birds. They maintain moisture levels in the banks that support the trees and shrubs, and prevent the riparian vegetation from being dominated by any one species. Floods also scour estuaries, ensuring, amongst other things, accessibility to marine fish dependent on them as nursery areas, and the maintenance of habitat diversity.
- Larger floods can be catastrophic and costly for both the river system and human life, however they are ecologically important as they reset the system, by altering physical and chemical conditions that influence the long-term development of biotic communities. Extreme droughts and floods are crucial for maintaining common biological, physical characteristics and ecological vitality in rivers (Naiman *et al.* 2008).
- Flow variability, on a daily, seasonal or annual basis, acts as a form of natural disturbance. This maintains biological diversity through increased heterogeneity of physical habitats. For instance, lack of variability through the absence of small floods may favour fish species adapted to breed under conditions of more constant discharge, with resulting alterations in the relative numbers of fish species and/or loss of native species. Variability in low-flows dictates the width of the vegetation belt along the water line, which protects the banks against erosion. A loss of variability results in a narrowing of this band because the lower portion is no longer regularly exposed or the upper portion regularly inundated. Variability in low-flows dictates the width of the vegetation belt along the water line, which protects the banks against

erosion. A loss of variability results in a narrowing of this band because the lower portion is no longer regularly exposed or the upper portion regularly inundated. Thus, together these flows affect channel configuration, habitat provision and a host of other biological processes (Swanson *et al.* 1982; Poff *et al.* 1997; Tuner 1998; Friedman and Auble 2000; Pettit *et al.* 2001; Naiman *et al.* 2005; Camporeale and Ridolfi 2006; Gurnell *et al.* 2011).

Naturally, a river exists in a state of dynamic equilibrium, able to respond to seasonal and annual fluctuations in climate because its species have different tolerance ranges and so differ in their abundances as conditions change (Brown and King 2002). Thus, at any time there is a mix of species that can cope efficiently with prevailing conditions, while other species may be present in lower numbers or surviving as, for instance, eggs, seeds or spores, until more suitable conditions occur. The mix of species and numbers of individuals present usually result, in the natural situation, in assemblages where no one species proliferates to "pest" proportions.

Thus, together these flows affect channel configuration, habitat provision and a host of other biological processes (Swanson *et al.* 1982; Poff *et al.* 1997; Tuner 1998; Friedman and Auble 2000; Pettit *et al.* 2001; Naiman *et al.* 2005; Camporeale and Ridolfi 2006; Gurnell *et al.* 2011). For instance, Ewart-Smith (2012) found that seasonal variations in the flow regime explained 95% of changes in periphyton biomass and community composition, thereby showing that this temporal variability is exerted even at the very base of the food chain.

The size and type of substrate also has bearing on the species able to inhabit a particular area, e.g., a relatively uniform substrate will be favorable for a particular type of community and be a disadvantage to the other communities (Allan 1995, King and Schael 2001). Allan (1995) reported decline of diversity and abundance of macroinvertebrates with stone sizes (cobbles). Resh and Rosenberg (1995) found that larger substrata hosted different species than those found on smaller substrata. Suspended sediment also plays a major role in aquatic systems (Bredenhand 2005), and typically high silting decreases diversity and growth (Chutter 1998). Flow parameters such as water velocity; water as a medium and current as a force strongly determine distribution of biota (Allan 1995). Riparian vegetation plays an important role in dynamics of aquatic body and influences processes and community of the system (Vannote *et al.* 1980).

Frissell *et al.* (1986) proposed a hierarchical framework for river habitat classification that emphasises the river's relationship to its basin across space and time; from micro habitats, riffles, a connection of pools, tributaries to an entire network of channels (Lorenz *et al.* 1997). Within this hierarchy, Petts (1994) described river systems as three dimensional systems, driven by hydrology and fluvial geomorphology; structured by ecosystem food-webs; characterized by spiraling processes (processing of organic matter along the river length); and dependent on change, such as changing flows, moving sediments and shifting channels. Several major and influential concepts have been proposed to explain how these interact to drive ecosystem functioning.

In line with this hierarchy, in South Africa, geomorphological river zones are identified predominantly on the basis of valley form and slope (Rowntree *et al.* 2000), which tend to drive many of the other features. The source zone is short and located in the most upland basin and has low gradient. The mountain head water stream zone is very steep with water falls, plunge pools, bedrock fall and cascades. The mountain stream zone has a steep gradient stream dominated by bedrock and boulders with cobble and coarse gravel in pools. The transitional zone is moderately steep; the streambed is dominated by bedrock and boulders and is within a semi-confined valley floor with plane bed, pools and riffles. The upper foothills zone is moderately steep with a bedrock-cobble bed dominated by pools and riffles. The lower foothill zone a fairly lower gradient and a river bed that is a mixture of sand and gravel; pools, riffles, sand bars in pools and presence of a floodplain are common. The lowland river zone as a flat gradient and flow is relatively slow with an alluvial fine bed channel, as a result there is often a meandering channel pattern with a floodplain.

Landscape ecology attempts to understand and explain the interactions between spatial heterogeneity (structure) and flow of material and organisms between the ecosystems (ecological processes and functions; Turner 1998; Li and Wu 2004; Turner 2005). The central notion of landscape ecology that is where things are located relative to other things can be extremely important for them and what happens to them (Wiens 2002). Landscape ecology is strongly oriented towards land-use planning, in particular ecosystem patterning to support the scope of nature conservation (Blaschke 2006). Thus, in addition to the ecological effects of the spatial patterning (Turner 1989), landscape ecology considers broad spatial scales in relation to the management for biodiversity conservation (after Hobbs 1997). It attempts to integrate human activities into the landscape patterns (Wu and Hobbs, 2002) and processes. The theories and concepts of landscape ecology have proved increasingly useful in addressing issues of habitat fragmentation, reserve design and resource management (Wiens 1992).

Within landscape ecology, rivers have mostly been dealt with as an element of the landscape (Wiens 2002). There are three main perceptions with regard to rivers as, or within, landscapes (Wiens 2002):

- the river is an internally homogeneous element contained within a broader terrestrial landscape;
- the river is connected with the surrounding landscape by a series of flows across the land-water boundary, or longitudinally down the river corridor;
- the river is a part of a landscape but is internally heterogeneous, and thus there it is a ‘landscape’ itself.

Wiens (2002) emphasizes that although the science of landscape ecology had mostly focused on ‘land’, there is much to learn from studies of rivers are they are ecosystems that are strongly influenced by their surroundings at multiple scales (Allan 2004).

Ecological processes in landscapes have been studied at a range of different spatial and temporal scales (Risser 1987 in Turner 1989; Wiens 2002) ranging from small (usually just within stream reaches of a few hundred meters) up to basin-scale studies and larger (Allan 2004). The scale at which a study is conducted affect the results obtained to answer a particular ecological question, thus changing the size of the study area usually results to a different outcome (Turner 2001).

Concepts of river ecosystem functioning attempt to explain and conceptualise ecological connectivity and biotic response as a function of physical stream structure at different spatial and temporal scales within the patchy hierarchy of the river system (Poole 2002), but all view flow as the main agent. Table 2.1 summarises the driving concepts of ecosystem functioning and processes in the longitudinal, lateral, vertical and temporal dimensions. Here, the concepts are listed according to the components of the lotic landscape that they address. However, since each new concept builds on those before it, they are discussed in chronological order.

Table 2.1 Driving concepts of river ecosystem functioning

Component	River ecology concept	Brief explanation	Reference
Basin	Natural flow regime concept	The natural flow regime determines and controls the functioning of the river system, with all low and high flow periods equally important.	Poff *et al.* (1997)
River system	Process domain concept	Geomorphic processes govern/drive the temporal patterns of disturbances that influence ecosystem structure and function (large scale- in a basin)	Montgomery (1999)
Channel	River continuum concept	A river system is linked longitudinally but functions differently with distance from its source, i.e. the uplands, middle reaches and lowlands	Vannote *et al.* (1980)
	Serial discontinuity concept	Disturbance of the conditions in a river system before and after a disturbance (natural/anthropogenic) are different.	Ward and Stanford (1983)
Riparian area and floodplain	Flood pulse concept	Annual floods exchanges materials (nutrients and sediments) laterally between the channel and the floodplain and flood pulses are responsible for biota functioning	Junk *et al.* (1989)
Hyporheos	Hyporheic corridor concept	There is a vertical exchange of materials and biota between a river channel and its bed	Standford and Ward (1993)

The river continuum concept (Vannote *et al.* 1980) focuses on undisturbed river and rivers with no floodplains (Lorenz *et al.* 1997) and is arguably the most well-known concept of lotic ecosystem functioning. It conceptualises rivers as longitudinally-linked systems forming a continuum (river continuum concept, Vannote *et al.* 1989), but with distinct changes their length that dictate how biota live; for instance the uplands, middle reaches and lowlands and foothills are different from each other (Rodriguez-Iturbe and Rinaldo 2001; Finlayson and McMahon 2004).

The serial discontinuity concept (Ward and Stanford 1983) addresses functional and structural responses to discontinuities and regulation in the river system continuum, either as a result of natural or man-made features (Lorenz *et al.* 1997). In a river basin discontinuities can be represented as boundaries between adjacent river segments that differ geomorphologically. Each tributary in the basin network also creates a gap in the expected pattern of downstream transitions (Poole 2002). Usually the biophysical conditions after a disturbance/discontinuity are highly variable and often very different from those before disturbance (Stanford and Ward 2001; Downs and Gregory 2014). Two things are taken into account within this concept: the 'discontinuity' distance over which the physical or biological variable is shifted in the up/downstream direction; and, the intensity of the discontinuance, described as the absolute change in the physical or biological variables as a consequence of the disruption (such as a dam).

The flood-pulse concept (Junk *et al.* 1989) focuses on the lateral dimension and was mostly designed mostly for large floodplain rivers (Lorenz *et al.* 1997). Geomorphological and hydrological conditions produce pulses in river discharge that are the principal driving force responsible for the occurrence and productivity of, and interactions between, biota in rivers and on floodplains. The flood-pulse concept explains that properties of the floodplain and riparian area are not determined by their position on the river longitudinally, but rather by the magnitude, duration and frequency of the floods they experience. The lateral exchange of water between river channel and the riparian/floodplain areas, together with the nutrient recycling has a direct impact on the presence and behaviours of plants and animals at any point in time. The extent of predictability of floods, in terms of their magnitude, duration, timing of occurrence and rate of rise and fall, determines how the animals and plants evolve in response. Predictable floods tend to promote the occurrence of plants and animals with specialist life history traits, whereas unpredictable flood pulses favour the evolution of generalists (Lorenz *et al.* 1997; Capon 2003; Naiman *et al.* 2005).

The hyporheic corridor concept (Ward and Standford 1993) focuses on the hyporheos. It highlights that, although not apparent to the naked eye, there is a vertical exchange of materials and biota between a river channel and its bed.

The natural flow regime concept holds that the natural flow regime of the river system determines and maintains the ecological integrity of a lotic ecosystem (Poff *et al.* 1997; Richter *et al.* 2003; Naiman *et al.* 2008). According the concept, the flow regime of a river system dictates the temporal changes that happen within the basin, and identifies five components of the flow regime that are important in this regard: timing; magnitude; frequency; duration; and, velocity. Daily and seasonal fluctuations in low-flows, small floods that occur every year and intermediate floods with occurrence intervals of two to five year are important for the maintenance river ecosystem structure (King *et al.* 2003a). Larger floods with occurrence intervals of 20 years or greater can be catastrophic for the system. Rising and falling water levels transport and deposit sediments and vegetation debris to downstream locations controlling the distribution and abundance of aquatic species (Poff *et al.* 1997; Goodson *et al.* 2001).

Humans inhabit the lotic landscape and, to a greater or lesser extent, have impacted on every structural and functional aspect of rivers (Lorenz *et al.* 1997; Turner 2001; Table 2.2). While by no means exhaustive, Table 2.2 provides an indication of the main human activities that influence change in lotic landscapes. Many studies have pointed to human actions as a principal threat to the ecological integrity of river ecosystems, impacting habitat, water quality, and the biota through numerous and complex pathways (Allan 2004; Aguiar 2005). Understanding the role of spatial and temporal variability on links between geomorphic and ecological processes is important for understanding the consequences of land-use change on the ecology of drainage basins (Montgomery 1999). Hydrologic, geomorphic and ecological responses to landscape disturbance depend on the type and extent of land-use practices (Poff 2002). Agriculture is the main land-use in most developed catchments, followed by urban areas, making urbanization the second major cause of river impairment (Paul and Meyer 2001; Allan 2004) even although urban land-use often covers a small fraction of the total basin area (Corbacho *et al.* 2003).

Agricultural, forestry and other farming practices exert a disproportionately large influence both proximately and over distance (Paul and Meyer 2001) on biotic communities and individual species. Farley *et al.* (2005) reported a decrease in annual run-off at 26 catchments following a change in land-use from natural shrub and grassland to forestation; run-off had reduced by 44% and 31% respectively. A similar case was also reported between 1956 and 1980 for Loess Plateau, Western China. Deciduous trees were used to forest 80% of the catchment while the natural grassland remained unchanged. Reduced run-off, volume and peak flow of storm runoff was shown, an estimated 32% cumulative runoff yield was reduced as a result of afforestation, with up to 50% reduction of annual runoff within a period of 15 years. Added to this, a significant trend was also observed that shows annual runoff reduction increases with the age of the trees planted (Huang 2003). The interplay of different land uses affects the quality of water that is drained by the river basin. Runoff from urbanized surfaces, wastewater, agricultural and industrial discharges result in increased loading of nutrients, metals, pesticides and other contaminants in rivers. Such changes result to reduced water quality and declines in the species richness of aquatic biota such as invertebrates, fish and plant communities (Paul and Meyer 2001; Butler and Davies 2004). Klein (1979) pointed out that the evidence to water quality impairment is first noticed when 12% of a river basin is impervious, but does not become severe until imperviousness reaches 30%. In the Western Cape, South Africa unnatural changes within the Berg River Basin including large-scale land-use/cover, modification of the natural flow regime, changes to water quality, siltation, alien plant invasion and introduction of alien fish species have been linked to the change in distribution, abundance and eradication of some indigenous fish species within the system (Clarke *et al.* 2009). Agricultural run-off and effluent along the Berg River have also been linked to the increase of inorganic nitrogen and phosphorus levels downstream, highest between Paarl and Hermon (de Villiers 2007).

Table 2.2 Human activities that have direct/indirect influences on the lotic landscape

Categories of human induced changes	Direct impact	Indirect impact	Reference
Urbanisation and industrialisation	Reduces permeation and increases runoff	Channel erosion, loss of biodiversity and deterioration of ecosystem services in the river	Decamps *et al.* 1988; Mol 2000; Wu 2008; Paul and Meyer 2001; Bledsoe and Watson 2001; Brookes 2009
Afforestation, deforestation, drainage systems, vegetation clearing, agriculture	Decreased surface runoff. Increase evaporative losses and reduce recharge	Soil-water deficit, increased erosion	Poff *et al.* 1997; Le Maitre *et al.* 1999; Huang 2003; Farley *et al.* 2005; Graf 2006
Dams, weirs and impoundments	Flow regulation by dams, sediment trapping, hydro-power generation, changes in downstream flow regime, conversion of lotic to lentic habitat	Embedded gravels, channel stabilisation and narrowing, channel erosion, reduced habitat for fish and macroinvertebrates, loss of riparian area and nutrient transfer, floodplain disconnection	Poff *et al.* 1997; Beck and Basson 2003; Graf 2006
Pumping from rivers and groundwater	Volume of water in the channel. Reduced recharge	Altered dominance of plant species dependent on certain flood regimes	Le Maitre *et al.* 1999; Poff *et al.* 1997
Infilling of wetlands and floodplains	Changes in runoff patterns, in particular the magnitude and duration of floods and lowflows. Loss of habitat	Loss of biodiversity, loss of function	Davies and Froend 1999; Heath and Platter 2010; Cossart *et al.* 2014
Channel bulldozing, sediment mining, navigation, berms, infilling, clearing riparian vegetation, species addition and removals	Pollution, change in runoff, change in river channel structure	Loss of biodiversity, Loss of functionality	Poff *et al.* 1997
Waste and sewage disposal, and other organic pollution	increased loads of nutrients, metals, pesticides, reduced oxygen, algal blooms, increased nutrient concentrations (N and P)	Groundwater contamination	Paul and Meyer (2001); Butler and Davies (2004); Polunin (2009)
Pesticides, herbicides and fertilizers, mining drainage, industrial pollution	Reduced water quality	Loss of fish, invertebrates, etc., loss of drinking water sources	Allan 2004; Broussard and Turner (2009); Zhang *et al.* 2010
Fisheries and other harvesting of resources	Altered abundances of harvested species. Loss of fish, invertebrates and other aquatic biota	Shift in community structure. A collapse in the food chain. Habitat destruction. Alien invasion	Alcala and Russ 1990; Blaber *et al.* 2000; Clark *et al.* 2009
Inter-basin transfers, exotic fish	Reduced flows in donating river and increased flows in receiving river translocation of species, change in water chemistry	Life cycle disruption, loss of sensitive species	Bunn and Arthington 2002; Poff *et al.* 1997; Snaddon and Davies 1998

Changes in land-use within a river basin may also impact on channel pattern; multiple channels often changed into single thread channels (Gregory 2006). Although river channel change (including enlargement, shrinkage and metamorphosis) is broadly known, due to their complex responses and historical contingencies it is not always possible to predict the nature and amount of change that will happen at a particular location (Schumm 1979; Phillips 2001; Gregory 2006). Paul and Meyer (2001) found that an increase in impervious surface cover (for instance a 10-20%) results to a twofold increase in runoff (Paul and Meyer 2001), which affects sediment transport and deposition and can result in channel widening and deepening (Bledsoe and Watson 2001; Brookes 2009). Between 1966 and 2000 river run-off and flood magnitude increased in the Los Penasquitos Creek, Southern California in response to increased in urbanization (9-37%) of its basin, which lead to previously braided river channels in the lower reaches of the basin combining into a single deep channel (White and Greer 2006). Channel erosion due to landscape changes can also lead to increased sediments in the lower reaches of the basin (Trimble 1997, in Bledsoe and Watson 2001). Robinson (1976, in Klein 1979), reported that streams draining developed river basins averaged twice the channel width of rural streams, thus landscape changes were shown to have affected river morphology.

Water-resource developments and land-use changes affect rivers' flow regimes, water chemistry and sediment and temperature regimes and, as a knock-on effect, their fauna and flora (Poff *et al.* 1997; Bunn and Arthington 2002; King *et al.* 2003a). The hydrologic effects of large dams in America were studied on 36 paired rivers reaches, one located upstream (regulated) and the other downstream (unregulated) of a dam. Comparison of the regulated and unregulated reaches showed that on average, large dams reduce annual peak discharges by 67% (in some individual cases up to 90%) and decrease the range of daily discharges by 64% (Graf 2006). In Ghana, Thorne *et al.* (2011) found that heavily polluted sites supported a poor fauna and were dominated by pollution tolerant taxa. Amongst others the general and mostly reported invertebrate responses to disturbance are: a decrease in diversity and abundance in response to toxins, temperature change, siltation, and organic nutrients, as well as increased abundances in response to inorganic and organic nutrients (Paul and Meyer 2001).

Considerable focus has been placed on how communities are structured and on identifying the main environmental factors that determine composition and abundance in lotic environments because this provides information for monitoring (Richards *et al.* 1993; Rosenberg and Resh 1993). Monitoring is the periodic or continuous collection of data (measured parameters) using consistent Environmental Protection Agency methods aquatic biomonitoring programmes have been implemented all over the world (Roux 1999), e.g., Australian National River Health Programme and the Rapid Bioassessment Protocols in the United States (Roux 1999; Ollis *et al.* 2006). Biomonitoring programmes are developed for different reasons including; to survey the general ecological state of aquatic ecosystems, to assess effect of impacts, and to detect long term environmental trends (Roux 1999). Typically, these biomonitoring programmes are based on use of a combination of physical, chemical, and biological parameters (Dallas and Day 1993; Rosenberg and Resh 1993; Roux *et al.* 1993; Bredenhand and Samways 2009). Rapid approaches to biomonitoring are adopted due to demand for quick, low cost and effective techniques (Rosenberg and Resh

1993; Dickens and Graham 2002; Ollis *et al.* 2006). They should not be seen as replacement for the traditional detailed and quantitative surveys, but rather as a precursor to these (Ollis *et al.* 2006). It is also important to realise that there will always be a compromise in the accuracy and precision of results through the use of rapid biomonitoring techniques (Chutter 1994; Chutter 1995; Chutter 1998; de Moor 2002).

More than half of aquatic biomonitoring programmes in use today rely to some extent on analysis of macroinvertebrate communities (Czerniawska-Kusza 2005; Uherek and Gouveia 2014). They are regarded reliable indicators of ecosystem ecological integrity because they reflect the cumulative effects of impacts acting on the ecosystem over time (Dallas and Day 1993; Chutter 1995; Rosenberg and Resh 1993). They are used as bio-indicators because they are generally abundant, have little mobility, are often used in studies to determine the quality of waters because of their high numbers, known pollution tolerances, limited mobility, wide range of feeding habits, varied life spans. There is a large array of indices developed based on macroinvertebrate community structure (Ollis *et al.* 2006). These include: South African Scoring System (SASS), which is a modified version of methods used by the Biological Monitoring Working Party Score System (ISO-BMWP 1980) was developed for river assessments in South Africa (Chutter 1994; Chutter 1995; Chutter 1998), Namibia Scoring System in Namibia (Palmer and Taylor 2004), Stream Invertebrate Grade Number Average Level Biotic Index for eastern Australia (Chessman 1995), Danish Stream Fauna Index developed for use in Denmark (Skriver *et al.* 2000), Okavango Assessment System for Botswana (Dallas 2009a) and Zambian Scoring System for rivers in Zambia (Lowe *et al.* 2013)

In South Africa, the River Health Programme (RHP) was developed and implemented with a purpose of providing overall ecological status of river systems (Roux 1997; Roux *et al.* 1999); incorporating concepts from other international models (Roux 1999). The RHP is mostly based on biological monitoring of fish, macroinvertebrates and riparian vegetation as these primary indicators characterise the response of the aquatic environment to disturbance. Some abiotic indicators, including geomorphology and habitat integrity, are also used in RHP. In 2016 the RHP was replaced by River Eco-status Monitoring Programme (REMP), which incorporates the South African Scoring System and the National Aquatic Ecostatus methods (www.dwa.gov.za).

Meaningful interpretations of monitoring results should be underpinned by consideration of the influence of factors at a wide spatial scale and over long periods (e.g., Committee on Watershed Management 1999; Thompson *et al.* 2001). River monitoring by the EPA includes monitoring physical, chemical, and/or biological condition of the river as well as specific basin characteristics, such as extent and nature of riparian corridor, wetlands and land-use, that may be related to observed ecosystem condition (Natural Research Council 1995). Data collected at the wider spatial scale are used to support interpretation of site specific biological monitoring such as the collection of algae, aquatic plants, aquatic insects, and/or fish, and to help establish cause-and-effect relationships.

Temporal scales also provide a framework for considering natural and human processes in watersheds. The history of an ecosystem's structure and functioning is important for understanding how present conditions came about (Turner 2005) as well as realising reference conditions (Newson 2008), and is also crucial to ecologically-sound management of rivers (Bis *et al.* 2000; Rhemtulla and Mladenoff 2007). This information can be used to identify and document past and current pressures on the system, establish the historical context for the aquatic ecosystems, enhance the understanding of how these responded to past pressures, and ensure that ongoing monitoring data are interpreted within an understanding of past pressures on the system (Committee on Watershed Management 1999).

3 Historical changes in land-use of the Berg River Basin

3.1 Introduction

Humans inhabit lotic landscapes and, in so doing, alter their characteristics and those of the rivers that drain them (Lorenz *et al.* 1997; Foley *et al.* 2005). These changes in natural landscapes as a result of human use are a major and on-going threat to biodiversity and ecosystem functioning. In river ecosystems, the riparian zone and floodplain serve as buffers between the channel and the surrounding lotic landscape. The condition of these areas strongly influences the hydrology, chemistry, temperature and physical characteristics of the rivers (Harding *et al.* 1998). Floodplains are highly attractive for human exploitation as they are among the most productive and diverse of ecosystems (Poff *et al.* 1997). As a case in point, in Japan 50% of the human population and more than 70% of the assets are located on 'former' floodplains (Nakamura *et al*. 2006).

The effects of human activities range from direct physical impacts, such as loss in riparian and aquatic biota to indirect consequences, such as a decline in water quality and exacerbating climate change (Forester *et al.* 2003, De Fries 2004). Agricultural, industrial or urban areas are created where once there were forests, shrublands and floodplains (Grau *et al.* 2003; Tockner *et al.* 2010). Urbanisation causes catchment hardening, severely altered flow regimes, and channelization or canalisation of rivers; this leaves little room for natural ecological functioning (Poff *et al.* 1997; Paul and Meyer 2001; Allan 2004; Wu 2008). Outside of the built up areas, clearing of vegetation for agriculture also changes the pattern and intensity of water flowing off the landscape into the river and often increases sediment supply to river channels, resulting in changes to inputs of organic matter and incident light, reduced river bank stability and ultimately loss of species and natural function (Poff *et al.* 1997). Other, less obvious but equally insidious, impacts of cultivation are homogenisation of top soil and depletion of carbon and nitrogen levels (Richter *et al.* 2000). Indeed, the many physical, chemical and biological changes imposed on soils by agriculture, commonly coupled with burning and grazing, means that the imprints of historic changes in land-use on soil properties are long-term and may have serious implications for the dynamic functioning of the ecosystems they affect (Davidson and Ackerman 1993; Trimble 1999).

Small scale changes in land-use are typically not considered as influential on river condition (Harding *et al.* 1998), but large-scale historical changes in land-use have shown considerable promise in explaining and understanding factors controlling lotic ecosystem condition (Goodale *et al.* 2000; Foster *et al.* 2003), although the results are not always as expected. For instance, where it would normally be expected that rivers draining agricultural environments support a lesser diversity of aquatic biota overall (Wang *et al.* 1997; Meyer *et al.* 1999; Allan 2004), a study of historical changes in land-use on the diversity of aquatic invertebrates and fish showed that while the abundance and diversity of macroinvertebrates were higher in forested basins, those of the fish were higher in agricultural basins (Harding *et al.* 1998).

The effects of human-induced global changes in land-use include habitat loss and fragmentation, soil degradation, species invasions and extinction, changes in vegetation type and river channel modifications (Lorenz *et al.* 1997; Harding *et al.* 1998, Grau *et al.* 2003; De Fries *et al.* 2004; Foley *et al.* 2004; DEAT 2006; Musvoto 2008; Chapin *et al.* 2011). According to Scheffer *et al.* (2001), human alterations of lotic landscapes may initially have little effect on the ecological condition of the rivers draining them, but as changes escalate through time, they have a profound knock-on effect on ecosystem resilience, which leaves rivers more vulnerable to abrupt and persistent changes to their structure and function; changes to the complex linkages and interactions that are responsible for shaping and sustaining a particular ecological system can lead to irreversible negative impacts (Harding *et al.* 1998; Tockner *et al.* 2010). Although some recovery and restoration of physical habitat is possible, the degree to which aquatic and terrestrial ecosystems recover from long-term large-scale change is still relatively unknown (Harding *et al.* 1998). Consequences of changes may continue to influence environmental conditions long after the initial changes are made (legacy effects), for example elevated concentrations of heavy metals in river sediments from hard-rock mining in the late 1800s were responsible for reduced abundances and diversity of native fish in Northern Idaho in the early 2000s (Maret and Maccoy 2002, cited by Allan 2004). A similar study in Puerto Rico demonstrated this legacy effect following restoration and recovery of a basin from large scale agricultural and manufacturing activities over a period of 40 years. Despite the surrounding forest recovering structurally, reaching mature levels for local biodiversity and biomass in about 40 years, ecosystem structure of the forest and function remained influenced by exotic species (Foster *et al.* 2003, Grau *et al.* 2003). The history of a river basin's lotic landscape has also been successfully used to determine and predict present-day ecological conditions of southern Appalachian rivers (Allan 2004). Here, the data showed that changes in historic land-use were the predominant influence on composition and diversity of macroinvertebrates and fish communities in the rivers. From studies such as these, there is a growing awareness that site-specific conditions are influenced by broader ecological processes operating at the scale of the river basin and that maintenance of biodiversity may thus require conservation efforts over an extended area, and if possible, over the entire river basin (Harding *et al.* 1998; Allan 2004). Coupled with this is the fact that ecological studies often require extrapolation of fine-scale (site specific) data to broader scaled analyses (Turner *et al.* 1989). This is often done without quantifying many potential sources of error as spatial and temporal scales are changed.

In South Africa, water resources were initially managed based on flow alone without considering the ecological and environmental processes (Palmer 1999) but in the last few decades this has progressed to include consideration of the ecosystems underpinning the water-resources and the use of biophysical data in an integrated approach that seeks to explain how physical drivers of river condition, such as water and sediment flows, geomorphology and riparian vegetation, influence the biological responses of aquatic macroinvertebrates, fish, birds, herpetofauna and semi-aquatic mammals, in an ecologically relevant way (DWAF 1998). In practical terms this has taken the form of setting (and now implementing) an Ecological Reserve for every significant water resource in the country; and monitoring the ecological condition of these water-resources (DWS 2016). In South Africa, the previous European based water law (Water Act 54 of 1956) was reviewed and replaced by the National Water Act 36 of 1998 which was introduced after the transition to a democratic government in 1994. During the apartheid government the old water act, legal rights and access to water (both ground and surface) were based on land ownership (Palmer 1999). The 1956 Act did not recognise water as a basic human right but ensured that access to water was mostly by a small dominant group with privileged access to land and economic power. After 1994, sociologists, ecologists and environmental scientists became involved in policy development and law making, which led to a water law reform that aimed at meeting political and social goals of equitable water access. The resultant key environmental principle was that the quantity and quality of water required to meet basic human needs and to maintain environmental sustainability will be guaranteed as a right. This is laid out in Chapter 3 Part 3 of the NWA Act of 1998; “the Reserve” which consists of two parts, the basic human needs reserve and the ecological reserve. The National Water Act of 1998 is the principal legal instrument relating to water resource management and the Department of Water Affairs and Forestry (DWAF) was responsible for the development and management of the country’s water resources in terms of NWA. Government objectives for water resources in South Africa are set out in the National Water Resources Strategy (NWRS) as to achieve: equitable access, sustainable use together with an efficient and effective water use. (i) To achieve equitable access to water: apart from water belonging to all people, the Act recognizes the importance of an equal access and use of water services and resources. (ii) To achieve sustainable use of water, by promoting water use that benefits the achievement of an equitable and sustainable social and economic development also ensure environmental protection. (iii) To achieve efficient and effective water use for optimum social and economic benefit. In support of this are other legislations that govern river basin activities such as the Water Services Act (Act 108 of 1997), National Environmental Management Act (NEMA, Act 107 of 1998) and Conservation of Agricultural Resources Act (CARA; Act 43 of 1983).

With this in mind, the objective of this chapter was to document large-scale land-use changes in the Berg River Basin over space and through time, analysing whether changes have taken place and if so, describing what these changes were. The central assumption was that all activities in the lotic landscape contribute either directly or indirectly to a river’s ecological condition, as defined broadly by physical, biological and chemical attributes (Harald *et al.* 1975; Dorava *et al.* 2001; Naiman *et al.* 2005).

Humans drive change across lotic landscapes as agricultural settlements expand to increase production of food and produce for enlarging and expanding development in urban areas as population growth and economic development increases (Grau *et al.* 2003; Tockner *et al.* 2010, www.westerncape.gov.za). Food production in agricultural areas has intensified driven by the need for higher yields and multiple crop types (Alexandratos 1999; Turner 1999). The Berg River is relatively small compared to other rivers in the Western Cape and has experienced an increase in population that is heavily dependent on agricultural production, which comes with high consumption of water resources. For these reasons, an analysis of historical changes across the lotic landscape of the Berg River Basin was expected to show that:

(1) Changes in land-use are progressive.
(2) The rate of change in land-use has accelerated over time.

3.2 Study area

The study area was the Berg River Basin in the Western Cape Province, South Africa, from its source to the maximum tidal influence of the sea at its estuary. The Berg River is one of the smaller rivers in the Western Cape (Ractliffe *et al.* 2007) and is approximately 285 km long from source to sea, with a basin area of approximately 9 000 km^2. It has its source in the Drakenstein and Franschhoek mountains, south of Franschhoek, about 6 km upstream of the Berg River Dam, which was completed in 2007. It flows northwards past the towns of Paarl, Wellington, Hermon and Gouda then turns west, passing Piketberg and Hopefield to the Atlantic Ocean on the West Coast at Velddrif. The Berg River Estuary is one of South Africa's largest estuaries with area about 61 km^2; based on the extent of its tidal influence, it is about 65 km long. The main tributaries of the Berg River are the Dwars, Franschhoek, Wemmershoek, Hugos, Krom, Kompanjies, Doring, Klein Berg, Sandspruit, Twenty-fours, Moorreessburg and the Sout Rivers (Figure 3.1). The basin lies in the winter rainfall area of the south-western Cape, with most of the rainfall (about 80%) received between April and September (Nitsche 2000).

The Berg River Basin experiences a Mediterranean climate with warm, dry summers and cool, wet winters (Moor and Day 2013). The basin comprises sequences of rocks of the Malmesbury Group, the Cape Granite Suite, the Klipheuwel Group, the Table Mountain Group and younger Cenozoic sediments in the western part of the basin (Ractliffe *et al.* 2007). The western part of the basin are underlain by unconsolidated sandy deposits and has few drainage channels; while the eastern and central parts, which are underlain by weathered and fractured rocks, have a higher drainage density. The Malmesbury and Klipheuwel Group comprise soft, erodible rocks that form flat plains in low lying areas, which generally have a moderate to high agricultural potential. The Berg River is a main source of domestic water supply for the City of Cape Town but it is also important for agricultural, industrial and environmental purposes (Leaner *et al.* 2012).

European settlement and farming in the upper Berg River Basin began in 1687, with the allocation of land on credit to Huguenot settlers in Drakenstein (later known as Paarl) and a few year's later in Olifantshoek (now Franschhoek) (Burman and Levin 1979), while there were some fishing and whaling activities in the lower basin around Saldanha intermittently from a slightly earlier time. Development was relatively slow until the end of the 19th century when agriculture progressively expanded into stock, wheat and fruit farming and, in some areas, rooibos tea (de Swardt 1983). After about 1930, favourable climatic and market conditions saw a surge in agricultural growth in the basin, accompanied by a growing urban element associated with the main towns: Franschhoek, Paarl, Wellington, Malmesbury, Tulbagh, Porterville, Piketberg and Laaiplek.

3.2.1 Geomorphological zones

In accordance with DWS's country-wide hydrological classification (DWAF 2007), the Berg River Basin is divided into 12 quaternary basins: G10A to G10M (Figure 3.2). Of these, Quaternary G10A, B and G are the most upstream and the smallest of the sub-basins, each less than 200 km^2; and G10K, L, M are the largest, with areas greater than 1750 km^2.

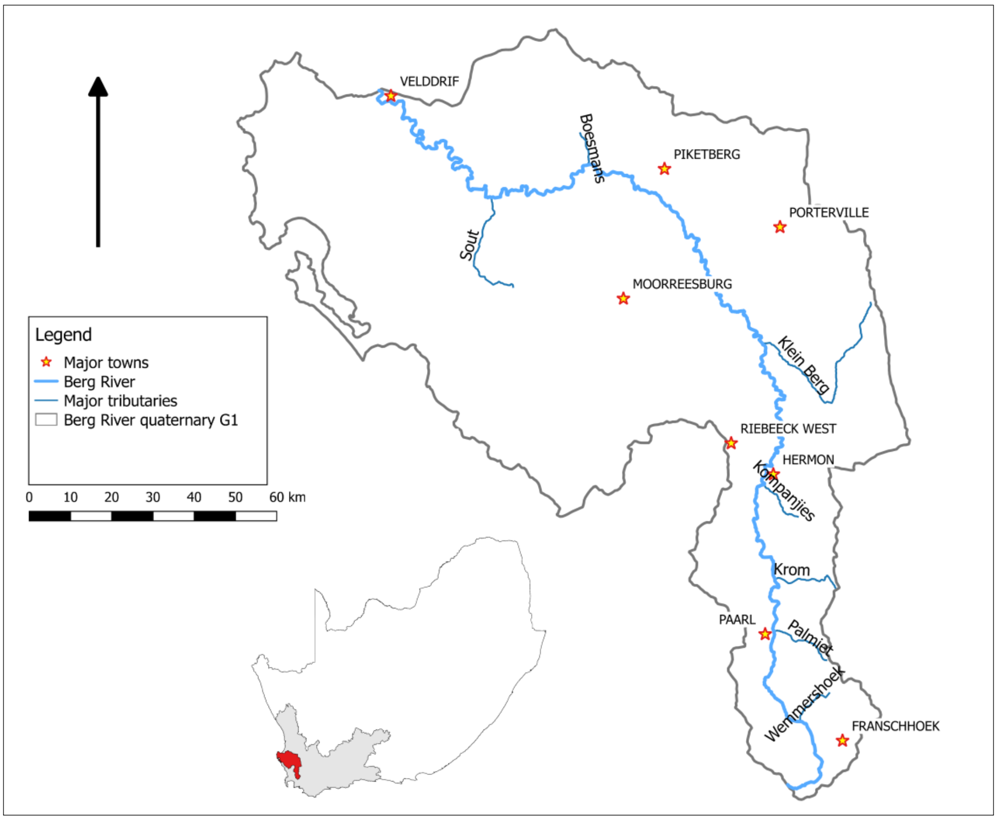

Figure 3.1 The Berg River and its main towns and tributaries. Insert shows a map of South Africa, the Western Cape Province (grey) and Berg River Basin G1 (red)

Six geomorphological zones, based on longitudinal slope (Rowntree *et al.* 2000), were delineated along the Berg River: the mountain headwater; mountain stream; transitional; upper foothill; lower foothill, and; the lowland zones (Figure 3.2 and Appendix Table 1); followed by the estuary. These zones were used to delineate sub-basins (see Table 3.1).

The mountain headwater, mountain stream, transitional and upper foothill zones are located in quaternary G10A (Ractliffe *et al.* 2007). Of these, the mountain headwater, mountain stream and transitional zones were short and were combined into a single sub-basin referred to here as the mountain stream sub-basin. This sub-basin contains the undeveloped reaches of the Berg River, upstream of the Berg River Dam. The upper foothills drain mostly minor tributaries ending just downstream the confluence of with the Franschhoek tributary (Figure 3.2). The lower foothill zone is located in quaternaries G10B to G10D; from the confluence with the Wemmershoek River to the confluence with the Kompanjies River. This sub-basin contains the now-extensive urban areas of Paarl, Dal Josafat, Wellington and settlement of Pniel. The lowlands flow through quaternary basins G10E-H and G10J-L between the Kompanjies and Sout Rivers. This sub-basin contains the settlements of Hermon, Gouda, Riebeek-West, Bridgetown, Mooreesburg, Hopefield and Piketberg. The lower most quaternary basin, G10M, comprises the estuary and coastal zone at Veldrift and Laaiplek, which are influenced by tidal effects; no major tributaries flow into the Berg River from this sub-basin (Figure 3.2).

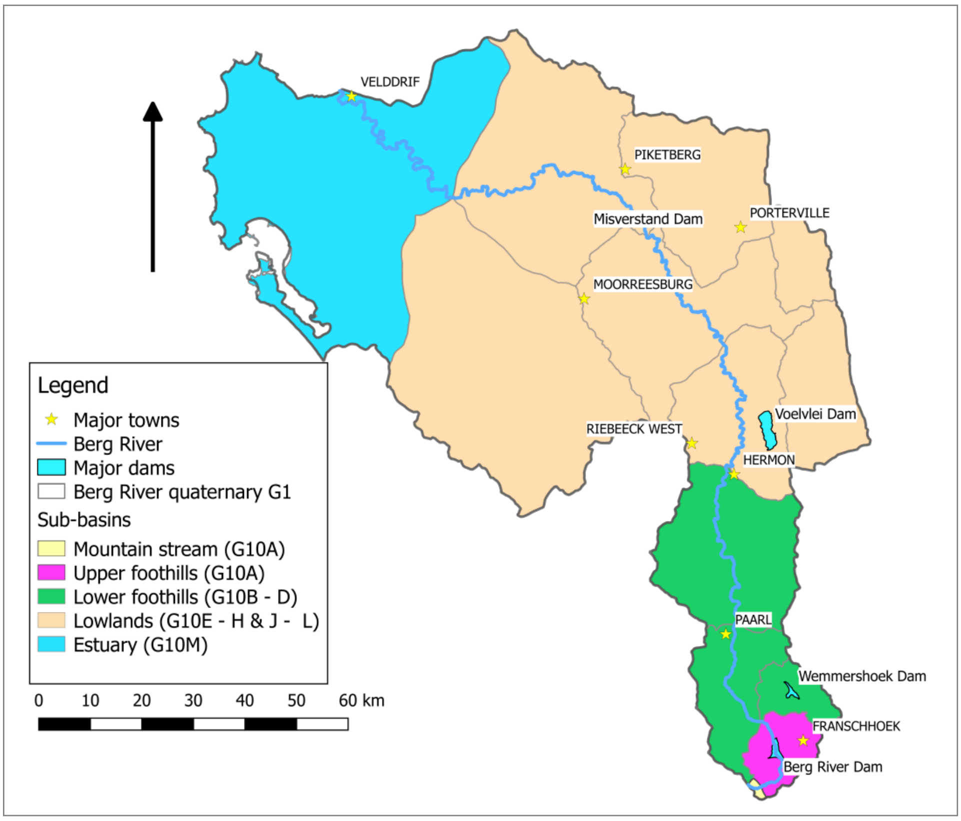

Figure 3.2 Geomorphological zones of the mainstem Berg River, and the sub-basins that feed each zone

The areas of the sub-basins vary widely (Table 3.1; Figure 3.2). The largest sub-basin drains into the lowlands and has an area of 5 621.82 km^2; followed by the estuary drainage area (2 015.72 km^2) and the lower foothills (1 147.51 km^2). The upper foothills and the mountain stream sub-basins are considerably smaller at 165.71 km^2 and 7.13 km^2 respectively.

Table 3.1 Geomorphological sub-basins of the Berg River Basin

Geomorphological sub-basin	Quaternary basin	Area (km^2)
Mountain stream	G10A	7.13
Upper foothills	G10A	165.71
Lower foothills	G10B-G10D	1 147.51
Lowlands	G10E-H & G10J-L	5 621.82
Estuary	G10M	2 015.72
Whole basin	8 957.71 km^2	

3.2.2 Land-use

The main land-use in the basin is agricultural crop production and there were approximately 600 farms in 2016 (Claassen 2015). Land-use is characterised by the rain-fed cultivation of wheat, extensive irrigation of vineyards and fruit orchards and ever-increasing development of urban areas (Basson and Rossouw 2003). Deciduous fruit (mainly grapes) is the backbone of the economy in the Berg River Basin and most major industries are agriculturally based and include wineries, canneries and other food-processing factories. Dominant crops irrigated in the upper and middle reaches were vineyards (51.6%), artificial pasture (10.6%), soft and tropical fruits (apricots, pears, peaches, oranges, lemons etc) (10.9%) and artificial pasture (10.6%); while the lower reaches were predominantly dryland farming (Quibell 1993). The areas surrounding the upper and lower foothill towns of Paarl, Wellington and Franschhoek are dominated by wine grape production; while dryland grain farming and sheep farming are predominant north of Wellington. The lowland towns of Malmesbury, Moorreesburg and Piketberg are major wheat-production centers, although some viticulture has extended into these areas in recent years (Pegram and Baleta 2014). Commercial pine forests used to occur in the upper reaches, near Franschhoek (EWISA 2007), but these were cleared in *circa* 2006 and management of the mountain stream sub-basin has been handed to conservation authorities.

3.2.3 Large dams

There are four large water-supply dams in the Berg River Basin: two on the Berg River, one on the Wemmershoek River, and the other an off-channel storage facility alongside the Berg River supplied by the Twenty-fours, Klein-Berg and Leeu rivers (Table 3.2).

The Berg River Dam impounds a portion of the foothills of the Berg River and is the most recent impoundment, commissiioned on 2008. It is used to meet agricultural demand for water in the summer growing season, a period of low flow (Gorgens and de Clercq 2005, Ractliffe *et al.* 2007). Water is also exchanged between the Berg and Theewaterskloof dams via an inter-basin transfer tunnel that passes through the Franschhoek Mountains. The Berg River Dam releases compensation water to supply irrigation demands as far as Zonquasdrift; downstream of Zonquasdrift, irrigation water is released from Voelvlei Dam (Gorgens and de Clercq 2005, Ractliffe *et al.* 2007).

Table 3.2 Large dams (> 5000 ML) within the Berg River Basin

Dam	Year completed	Capacity (ML)	Location	Main supply river
Wemmershoek	1957	58 644	Wemmershoek	Wemmershoek River
Voelvlei	1971	164 000	Gouda	Klein Berg, Leeu and Twenty-fours river
Misverstand	1977	7 737	Piketberg	Berg River
Berg River	2008	130 000	Franschhoek	Berg River

The Wemmershoek Dam is located on the Wemmershoek River, the confluence of which with the Berg River defines the boundary between the upper foothill and the lower foothill sub-basin (Figure 3.2). Wemmershoek Dam is used to supply Cape Town, but releases are occasionally made to supply downstream irrigation demands.

Voelvlei Dam is an off-channel storage facility in the lower foothill sub-basin (Figure 3.2) that was the first large water-supply scheme developed on the Berg River. It was completed in 1952 (Ractliffe *et al.* 2007, Brown and Magoba 2009). It is currently supplied by diverting runoff from the Klein Berg, Twenty-fours and Leeu rivers through a system of canals.

Misverstand Dam is located on the Berg River downstream of Voelvlei Dam, in the lowland sub-basin. It supplies water to Moorreesburg, Vredenburg, Saldanha Bay and Langebaan.

With the exception of Misverstand Dam (Table 3.2), these dams are part of the inter-linked Western Cape Water Supply System (Collins and Herdien 2013), which supplies water to Cape Town and surrounding areas.

3.3 Methods

3.3.1 Data collation

Data on land-use in different periods were collated to enable description of large-scale land-use changes in the lotic landscape through space and time. This was done by digitising land-use from a series of historical 1:50 000 topographic maps to capture data on proportional changes in land-use over time. Nine land-use classes were considered and grouped into four main categories; agricultural lands, urban areas, buildings outside of urban areas and water bodies (Table 3.3). Some of these were captured as polygons of area while others captured as were point data (see below). The three agricultural land-use classes, captured in area were: (1) dryland farming; (2) orchards and vineyards, and; (3) plantations. The two urban land-use classes, captured as areal extent, were (1) towns and (2) townships. There were four land-use classes captured as point data for buildings outside of urban areas: (1) farms; (2) industrial buildings; (3) towns, and; (4) townships. The four land-use classes of waterbodies captured as point data were: (1) dams; (2) perennial pans; (3) non-perennial pans, and; (4) dry pans.

The data were captured from 1:50 000 topographic maps obtained from the National Geo-spatial Planning Office at the Department of Rural Development and Land Reform in Mowbray, Cape Town (Table 3.4).

A full basin cover comprises 25 maps at 1:50 000 scale maps: grid reference indexes 3217, 3218, 3318, 3319 and 3419. Near to full coverage of the basin was only available for four time periods (Table 3.4). This was revealed through an analysis of the coverage of all topographical maps available (Appendix Table 2). Maps 3318ac, 3318da and 3319ad, which are mountainous and undeveloped areas located on the periphery of the study area corner edges of the basin, were missing for all the periods. The data layers were grouped into 10-year periods to maximize use of available maps. In this study, these 10-year periods are treated as a single point in time. The periods of 1955-1965, 1976-1985, 1996-2005 and 2006-2015 were selected on the basis of available maps for analysis (Table 3.4). Periods denoted are approximate, actual dates are given in the table. Some of these periods had two or three missing maps (bold on Table 3.4), which meant the land-use for these time periods needed to be patched.

To patch gaps, a map of the same location from the next closest period before and after were used to calculate the proportional annual change in extent of the different land-uses, and this was used to interpolate area to the year where the map was missing (Table 3.5). This process is based on the (untested) assumption that change was linear over time; by calculating an annual increment of change that was then multiplied by the appropriate number of years to fill the gaps (actual values presented in Appendix Table 3). The linear annual rate of change calculated from map 3319ac was used to patch 3319aa because the two maps are adjacent to one another so it was presumed the changes were happening at similar rates.

Table 3.3 Definitions of land-use classes

Category	Land-use class	Description	Data type
Agricultural lands	Dryland farming	Agricultural land used for ploughing mainly wheat and cash crops	Area (km^2)
	Orchards and vineyards	Irrigated land used for orchards and vineyards	
	Plantations	Includes stands of pine and other forested land, without differentiating between native forests and invasive aliens. Being unable to differentiate the two was a limitation to analysis	
Urban areas	Towns	The spatial extent of the built up and paved areas of main towns	Area (km^2)
	Townships	The spatial extent of the built up and paved areas of townships	
	Towns	The number of built up and paved areas of main towns	Point data (counts)
Buildings outside of urban areas	Farms	Building structures that are not necessarily within a town and are often on farm lands. They include residential settlements, farm houses, small farm villages and estates	Point data (counts)
	Industrial buildings	Buildings such as factories, lime and salt-works, and abattoirs	
	Towns	The number of the built up and paved areas that are towns	
	Townships	Informal residential settlements and villages smaller than towns	
Water bodies	Dams	Water bodies with a wall on at least one side	Point data (counts)
	Non-perennial pan	Water bodies that are empty during some part of the year	
	Perennial pan	Water bodies with no wall that have water throughout the year	
	Dry pans	Stand-alone man-made areas without walls or water	

Maps were vectorised in Quantum GIS (QGIS) to facilitate capture of the extent of each land-use class in each 10-year period. Polygons for agricultural lands and urban areas were digitised by creating vertices on the outer boundary of the different land parcels to capture the extent (km^2) of dryland farming (mainly wheat), orchards and vineyards, plantations and urban land-use classes. Shapefiles for urban areas were created for main towns and townships; the outskirts were digitized to obtain the urban area per period. For point data (counts), a single point was created to represent the location of buildings outside of urban areas; this layer is categorized into four classes: towns, townships, farms and industrial buildings. Points were also created for waterbodies; they were sub-divided into dams, perennial pans, non-perennial pans and dry pans.

Table 3.4 Dates of topographic maps used for each period, with bold values indicating years that were patched

Map index	Time period			
	1955-1965	1976-1985	1996-2005	2006-2015
3217 db_dd	1964	1980	2003	2006
3218 ca_cc	1965	1981	2003	2010
3218 cb_ca	1964	1986	2003	2006
3218 cd	1965	1982	2003	2010
3218 da	1964	1986	2003	2007
3218 dc	1961	1975	2003	2010
3218 dd	1961	1975	2003	2007
3219 cc	1963	1986	2003	2010
3317_3318 aa	1966	1981	1998	2010
3318 ab	1967	1980	2003	2010
3318 ac	-	-	-	-
3318 ad	1966	1981	1999	2010
3318 ba	1966	1980	2000	2010
3318 bb	1963	1981	2000	2010
3318 bc	1966	1979	2000	2010
3318 bd	1963	1978/1988	2000	2010
3318 da	-	-	-	-
3318 db	1963	1977	2000	2010
3318 dd	1959	**1975**	2000	2010
3319 aa	**1960**	**1980**	1997	2010
3319 ac	**1960**	**1980**	1997	2010
3319 ad	-	-	-	-
3319 ca	1958	1979	1997	2010
3319 cc	1962	1977	1997	2010
3419 aa	1963	1979	1997	2010

3.3.2 Data analysis

Changes in land-use were quantified for the basin as a whole and then per sub-basin (Figure 3.2 and Table 3.1). To address the first hypothesis, *viz*.: that changes in land-use are progressive; it was necessary to demonstrate that change had taken place in land-use between the four time periods in a continuously positive direction. This was done by comparing the total area (of agricultural lands and built-up areas) and counts (of water bodies and buildings) for all land-use classes (Table 3.3) as percentages of the total area in each case. Maps and tables of the areal extent, and counts, of each land-use were produced and compared between the time periods to quantify change. To test the second hypothesis, *viz*.: the rate of land-use change has accelerated over time; rates of change for each time period were calculated by dividing the difference in area between two time periods by the number of years over which changes were evident. These rates were compared using a bar graph.

To test whether variables of different land-use categories were statistically different an analysis of variance ANOVA (Dunn and Clarke 1987) was carried out.

Table 3.5 Data used to patch gaps; data from 3319ac and 3318dd were used to construct data for 3319aa

Land-uses (km^2)		3319ac				3319aa				3318dd		
		1945	1960	1971	1980	1945	1960	1971	1980	1959	1975	1992
Agricultural	Dryland farming	244.6	249.3	252.8	255.6	79.8	81.3	82.5	83.4	28.1	19.9	11.3
	Orchards and vineyards	4.2	18.7	29.3	38.0	8.3	36.9	57.9	75.1	63.7	95.5	129.3
	Plantations	40.3	41.8	42.9	43.8	0	0.00	0	0	19.2	14.8	10.2
	Total agricultural land	**2234.2**	**2269.9**	**2296.1**	**2317.5**	**2033.1**	**2078.3**	**2111.4**	**2138.6**	**2070.1**	**2105.3**	**2142.8**
Urban	Towns	0.9	0.9	0.8	0.8	0.7	0.4	1	1.9	1.2	2.2	3.3
	Townships	0	0	0	0	0	0	0	0	0.2	0.4	0.7
	Total urban area	**0.9**	**0.9**	**0.8**	**0.8**	**0.7**	**0.4**	**1**	**1.9**	**1.4**	**2.7**	**4.0**
Infrastructure count												
Buildings outside urban	Farm buildings	57	63	68	71	14	50	54	56	97	97	97
	Industrial building	2	2	2	2	6	1	1	1	1	1	1
	Towns	2	2	2	2	1	1	1	1	1	1	1
	Townships	5	6	7	7	47	19	17	14	10	9	8
Waterbodies	Farm dam	25	112	176	228	127	153	173	188	40	93	151
	Perennial pan	0	0	0	0	0	0	0	0	0	0	0
	Non-perennial pan	3	1	0	0	0	0	0	0	00	0	0
	Dry pan	0	0	0	0	0	0	0	0	0	0	0
	Water bodies total	**28**	**113**	**176**	**228**	**127**	**153**	**173**	**188**	**40**	**93**	**151**

Change in the area of each land-use class was tested between different time periods for the whole basin and then for each sub-basin. Land-use data were grouped using four variable attributes; feature type, sub-basin, area (km^2) and time period. Changes in land-use were analysed for the two main categories: agricultural lands and urban areas. A Kruskal-Wallis test was applied post-hoc using Statistica V13 (StatSoft 2011) and the least square means were plotted on a graph. Changes in land use shown to be different were tabulated and summarised graphically using bar graphs.

3.4 Results

3.4.1 Changes in land-use over time

The distribution of agricultural land-uses across the basin at different time periods is shown in Figure 3.3 and quantified for the basin overall and each sub-basin in Table 3.6.

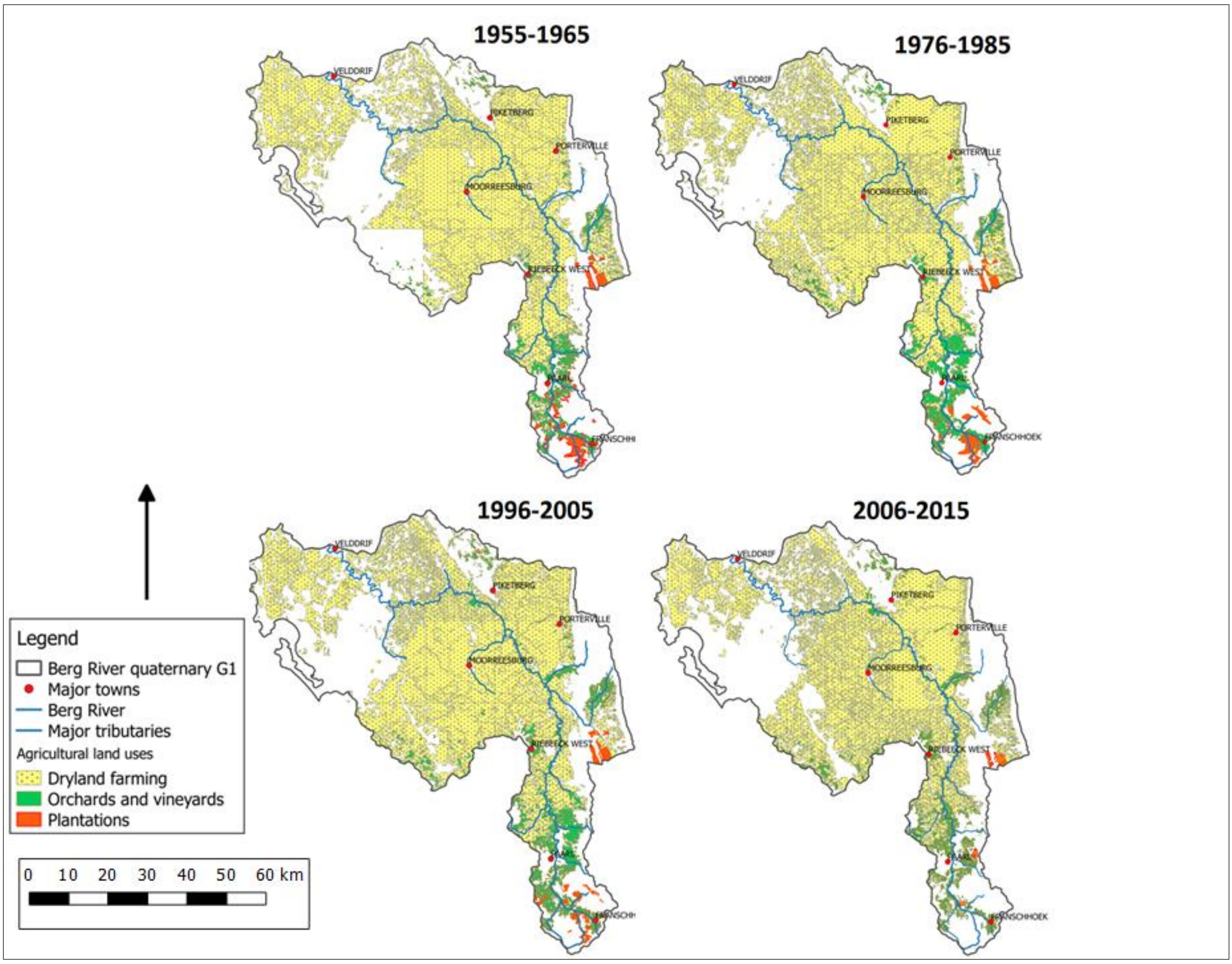

Figure 3.3 **Changes in agricultural land-use classes over time. Top left = land-use classes for 1955-1965, top right = land-use classes for 1976-1985, bottom left = land-use classes for 1996-2005 and bottom right = land-use classes for 2006-2015**

The areas of various agricultural land-uses for the whole basin differed between each time period (Table 3.6, Table 3.7 and Figure 3.4), many showing an initial increase, followed by a decline. Dryland farming did not differ much from one period to the next, but declined by 12% (±790 km^2) from 1955-1965 to 2006-2015 (Table 3.7). Orchards and vineyards increased overall by 6% (±32 km^2), and plantations decreased by 73% (±94 km^2). Total agricultural lands decreased by a similar amount to dryland farming from 1955-1965 to 2006-2015, by about 13% (±854 km^2).

Table 3.6 Area (km^2) and percentage of total basin area (%) of each land-use per sub-basin. Differences between neighbouring periods are shaded

Sub-basin	Land-use class	1955-1966		Difference between period period 1955-1965 and 1976-1985		1976-1985		Difference between period 1976-1985 and 1996-2005		1996-2005		Difference between period 1996-2005 and 2006-2015		2006-2015	
		km^2	% (basin)	Area	% in/decrease	km^2	% (basin)	Area	% decrease	km^2	% (basin)	Area	% decrease	km^2	% (basin)
Upper foothills (km^2)	Dryland farming	0.74	0.01	-0.20	0.00	0.54	0.01	0.68	0.01	1.22	0.01	0.00	0.00	1.22	0.01
	Orchards and vineyards	26.730	0.30	10.78	0.12	37.51	0.42	-6.05	-0.07	31.46	0.35	-9.61	-0.11	21.85	0.24
	Plantations	1.300	0.01	61.99	0.69	63.29	0.71	-33.94	-0.38	29.35	0.33	-29.35	-0.33	0.00	0.00
	Total agricultural land	28.77	0.32	72.56	0.81	101.33	1.13	-39.30	-0.44	62.03	0.69	-38.96	-0.43	**23.07**	**0.26**
	Undeveloped land	134.99	1.51	-72.13	-0.81	62.86	0.70	39.11	0.44	101.97	1.14	37.43	0.42	139.41	1.56
	Towns	1.40	0.02	0.07	0.00	1.47	0.02	0.23	0.00	1.70	0.02	-0.19	0.00	1.51	0.02
	Townships	0.55	0.01	-0.50	-0.01	0.05	0.00	-0.05	0.00	0.00	0.00	1.72	0.02	1.72	0.02
	Total urban	1.95	0.02	-0.43	0.00	1.52	0.02	0.18	0.00	1.70	0.02	1.53	0.02	3.23	0.04
Lower foothills (km^2)	Dryland farming	500.42	5.59	-41.29	-0.46	459.13	5.13	-58.49	-0.65	400.64	4.47	-47.83	-0.53	352.81	3.94
	Orchards and vineyards	280.65	3.13	108.41	1.21	389.06	4.34	-20.29	-0.23	368.77	4.12	-105.04	-1.17	263.73	2.94
	Plantations	41.98	0.47	-2.00	-0.02	39.98	0.45	-13.36	-0.15	26.62	0.30	-16.76	-0.19	9.86	0.11
	Total agricultural land	823.05	9.19	65.11	0.73	888.16	9.91	-92.13	-1.03	796.03	8.89	-169.63	-1.89	**626.40**	**6.99**
	Undeveloped land	306.68	3.42	-67.05	-0.75	239.63	2.68	69.76	0.78	309.40	3.45	163.43	1.82	472.82	5.28
	Towns	13.50	0.15	-2.26	-0.03	11.24	0.13	19.06	0.21	30.30	0.34	10.05	0.11	40.36	0.45
	Townships	4.28	0.05	4.20	0.05	8.48	0.09	3.30	0.04	11.78	0.13	-3.85	-0.04	7.93	0.09
	Total urban	17.78	0.20	1.94	0.02	19.72	0.22	22.36	0.25	42.08	0.47	6.20	0.07	48.28	0.54

Sub-basin	Land-use class	1955-1966		Difference between period period 1955-1965 and 1976-1985		1976-1985		Difference between period 1976-1985 and 1996-2005		1996-2005		Difference between period 1996-2005 and 2006-2015		2006-2015	
		km^2	% (basin)	Area	% in/decrease	km^2	% (basin)	Area	% decrease	km^2	% (basin)	Area	% decrease	km^2	% (basin)
Lowlands (km^2)	Dryland farming	4928.16	55.01	170.35	1.90	5098.51	56.92	-256.69	-2.87	4841.82	54.05	-348.26	-3.89	4493.56	50.16
	Orchards and vineyards	210.14	2.35	13.56	0.15	223.70	2.50	22.56	0.25	246.26	2.75	15.97	0.18	262.23	2.93
	Plantations	86.44	0.96	-42.48	-0.47	43.96	0.49	0.11	0.00	44.07	0.49	-19.08	-0.21	24.99	0.28
	Total agricultural land	5224.74	58.33	141.43	1.58	5366.17	59.90	-234.01	-2.61	5132.15	57.29	-351.31	-3.92	**4780.84**	**53.37**
	Undeveloped land	379.81	4.24	-139.40	-1.56	240.41	2.68	224.28	2.50	464.69	5.19	366.20	4.09	830.89	9.28
	Towns	15.38	0.17	-3.05	-0.03	12.33	0.14	7.27	0.08	19.60	0.22	-17.14	-0.19	2.46	0.03
	Townships	1.90	0.02	1.02	0.01	2.92	0.03	2.46	0.03	5.38	0.06	2.25	0.03	7.63	0.09
	Total urban	17.28	0.19	-2.03	-0.02	15.25	0.17	9.73	0.11	24.98	0.28	-14.89	-0.17	10.09	0.11
Estuary(km^2)	Dryland farming	1142.54	12.75	-31.86	-0.36	1110.68	12.40	-129.59	-1.45	981.09	10.95	-47.30	-0.53	933.79	10.42
	Orchards and vineyards	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.01	1.09	0.01	0.26	0.00	1.35	0.02
	Plantations	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Total agricultural land	1142.54	12.75	-31.86	-0.36	1110.68	12.40	-128.49	-1.43	982.18	10.96	-47.05	-0.53	**935.14**	**10.44**
	Undeveloped land	863.94	9.64	24.06	0.27	887.99	9.91	114.47	1.28	1002.46	11.19	28.38	0.32	1030.84	11.51
	Towns	7.17	0.08	8.35	0.09	15.52	0.17	9.99	0.11	25.51	0.28	10.16	0.11	35.66	0.40
	Townships	2.08	0.02	-0.55	-0.01	1.53	0.02	4.04	0.05	5.57	0.06	8.51	0.10	14.08	0.16
	Total urban	9.25	0.10	7.80	0.09	17.05	0.19	14.03	0.16	31.08	0.35	18.67	0.21	49.74	0.56
TOTAL	**Agricultural land**	**7219.10**	**80.59**	**247.24**	**2.76**	**7466.33**	**83.35**	**-493.93**	**-5.51**	**6972.40**	**77.84**	**-606.94**	**-6.78**	**6365.45**	**71.06**
	Urban	**46.26**	**0.52**	**7.28**	**0.08**	**53.54**	**0.60**	**46.30**	**0.52**	**99.84**	**1.11**	**11.52**	**0.13**	**111.36**	**1.24**
	Undeveloped land	**1685.41**	**18.81**	**-254.52**	**-2.84**	**1430.90**	**15.97**	**447.63**	**5.00**	**1878.53**	**20.97**	**595.43**	**6.65**	**2473.96**	**27.62**

At the basin level, the areas of orchards and vineyards, plantations, and urban areas of one period were all significantly different from the areas of the following period (Figure 3.4). Dryland farming at the basin level also differed from period to period, apart from the first two periods (1955-1965 and 1976-1985).

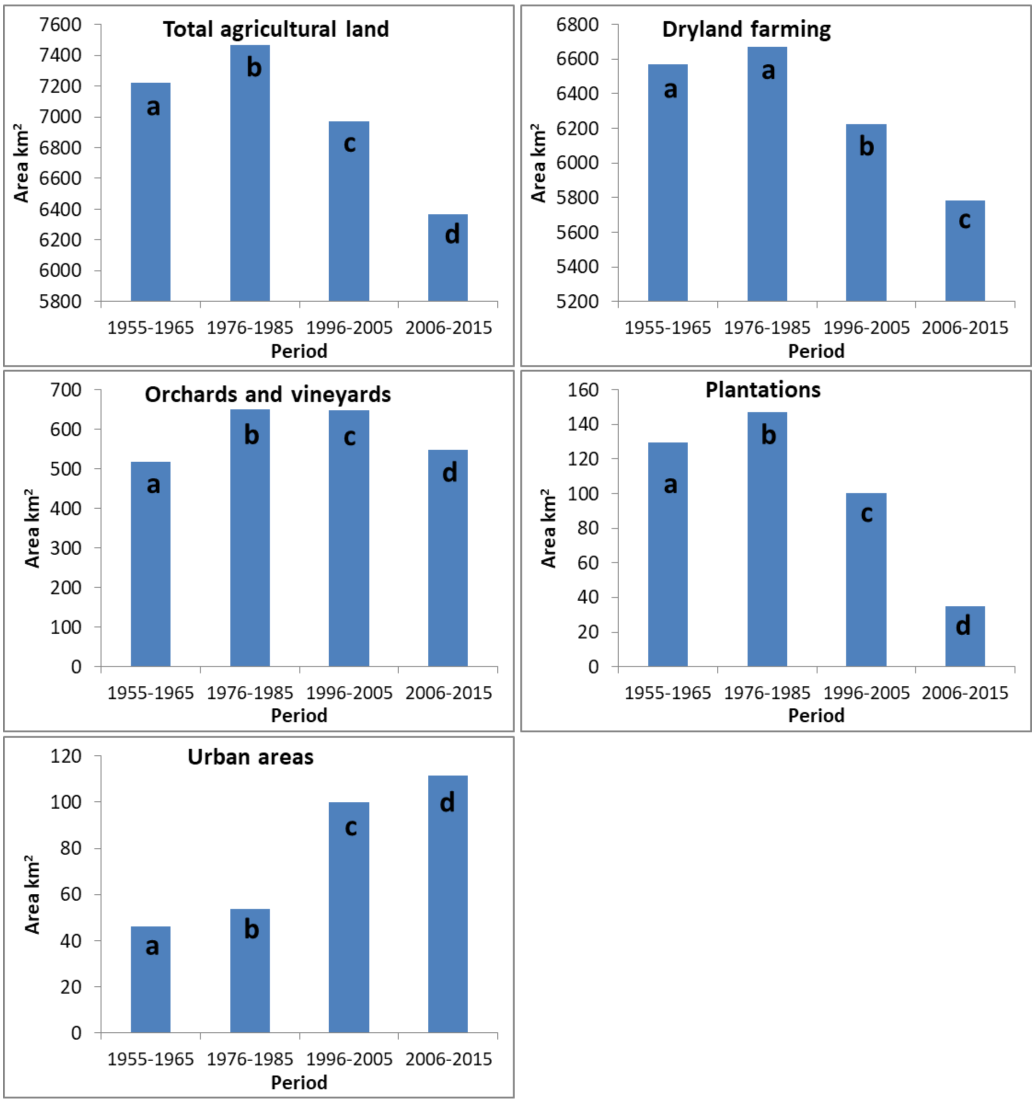

Figure 3.4 **Area of total agricultural land, dryland farming, orchards and vineyards, plantations and urban areas for the whole basin at different time periods. Letters differing between periods indicate significant differences using the Kruskal-Wallis test ($p<0.05$) (e.g. top left a and b) and where they are the same there is no significant difference (e.g. top right a and a)**

Total agricultural land also differed significantly from one period to the next (Figure 3.4). Figure 3.5 shows the ANOVA analysis of the basin level total agricultural land-use for each period. The reduction in agricultural land is the largest change (9.5%) between 1955-1965 and 2006-2015. The highest proportion of this loss was observed in the lowlands, with the loss of agricultural land here being 4.9% of the total basin agricultural lands, then the estuary

(2.3%), then the lower foothills (2.2%), while the upper foothills (0.06%) lost the lowest proportion (Table 3.7). The biggest portion of the loss in agricultural land was due to reductions in dryland farming (8.8%). Undeveloped lands increased by 8.8%, the highest part of which was in the lowlands (5.0%), followed by the lower foothill and estuary (1.8%) while the upper foothills (0.05%) had the least (Table 3.7).

Table 3.7 Changes in land use between 1955-1965 and 2006-2015, shaded show a decline. For waterbodies only dams are shown. Changes in the other water bodies can be found Appendix Table 8

Sub-basin	Land-use class	Difference 1955-1965 to 2006-2015	
		Area (km²)	Percentage of 1955-1965 area
Upper foothills (km²)	Dryland farming	0.48	0.01
	Orchards and vineyards	-4.88	-0.05
	Plantations	-1.30	-0.01
	Total agricultural land	-5.70	-0.06
	Towns	0.11	0.001
	Townships	1.17	0.01
	Total urban area	1.28	0.01
	Undeveloped land	4.42	0.05
	Water bodies (dams)	118	14.11
Lower foothills (km²)	Dryland farming	-147.61	-1.65
	Orchards and vineyards	-16.92	-0.19
	Plantations	-32.12	-0.36
	Total agricultural land	-196.65	-2.20
	Towns	26.86	0.30
	Townships	3.65	0.04
	Total urban area	30.50	0.34
	Undeveloped land	166.14	1.85
	Water bodies (dams)	647	48.83
Lowlands (km²)	Dryland farming	-434.60	-4.85
	Orchards and vineyards	52.09	0.58
	Plantations	-61.45	-0.69
	Total agricultural land	-443.90	-4.96
	Towns	-12.92	-0.14
	Townships	5.73	0.06
	Total urban area	-7.19	-0.08
	Undeveloped land	451.08	5.04
	Water bodies (dams)	879	96.53
Estuary(km²)	Dryland farming	-208.75	-2.33
	Orchards and vineyards	1.35	0.02
	Plantations	0.00	0.00
	Total agricultural land	-207.40	-2.32
	Towns	28.49	0.32
	Townships	12.00	0.13
	Total urban area	40.49	0.45
	Undeveloped land	166.91	1.86
	Water bodies (dams)	9	1.08

Sub-basin	Land-use class	Difference 1955-1965 to 2006-2015	
		Area (km^2)	Percentage of 1955-1965 area
TOTAL	**Agricultural land**	-853.64	-9.53
	Dryland farming	-790.48	-8.8
	Orchards and vineyards	31.64	0.35
	Plantations	-94.87	-1.05
	Urban area	65.10	0.73
	Undeveloped land	788.54	8.80
	Water bodies (dams)	**1727**	**130.34**

Overall, the total number of water bodies has increased across all sub-basins. Between 1955 and 2015, the highest increases in the number of dams were shown at the lowlands (96.53%) and lower foothills (48.83%) (Table 3.7).

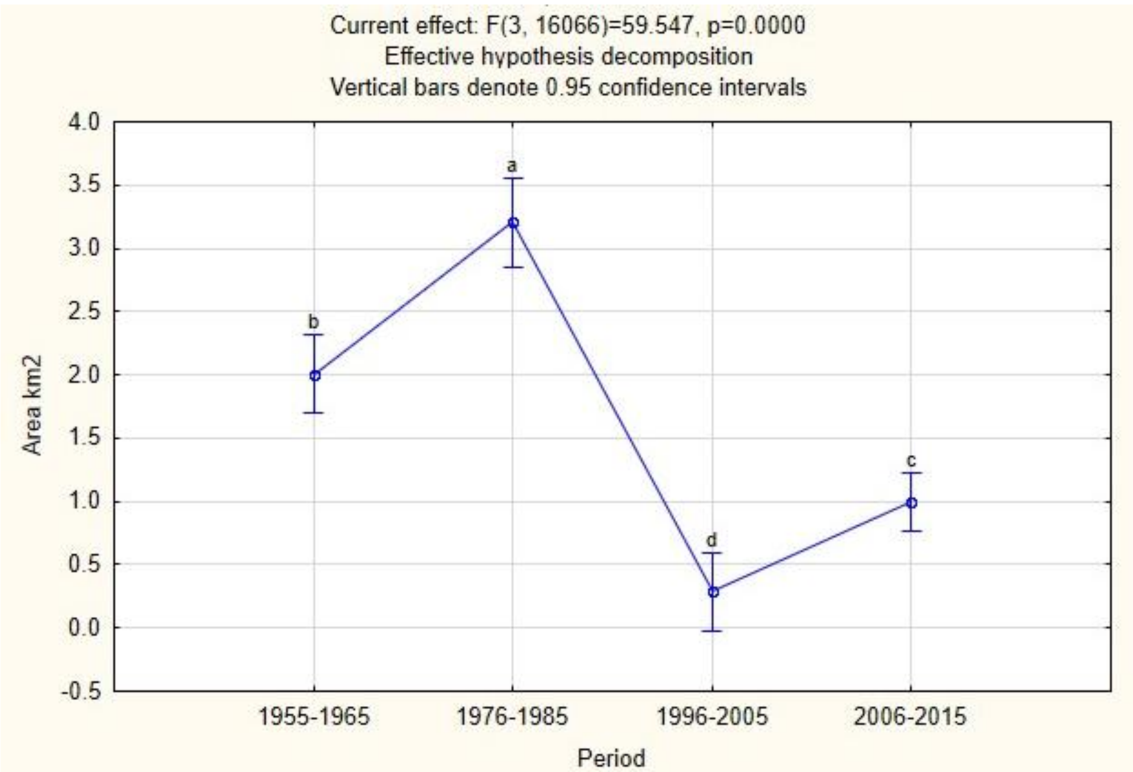

Figure 3.5 ANOVA for differences in distribution of total agriculture land-use through time. Letters differing between periods indicate significant differences (p<0.05)

The extent of urban areas (Appendix Table 8) increased progressively by ±7 km^2 (from 1955-1965 to 1976-1985), then ±46 km^2 (from 1976-1985 to 1996-2005), and ±11 km^2 (from 1996-2005 to 2006-2015). Over the whole period, the extent of urban areas across the whole basin increased by 0.7% (±65 km^2), with the biggest changes being in the estuary (Veldrift and Laaiplek), with a 0.4% increase, and in the lower foothills (Paarl and

Wellington), with a 0.3% increase. Of this, informal townships increased by > 0.2% in all sub-basins (Appendix Table 8).

The area of total agricultural land also differed over time in each sub-basin (Table 3.6 and Figure 3.6). In the upper foothills, this was driven by changes in the area of plantations and orchards and vineyards which increased by 62% from 1955-1965 to 1976-1985 but then declined back to nothing again by 2006-2015. This was accompanied by an overall 20% decline in total agricultural land in the upper foothills over the whole period. In the lower foothills, there was an initial 39% increase in orchards and vineyards at the expense of dryland farming and previously undeveloped land, but a subsequent decline to levels similar to those in 1955-1965. There was an overall 24% decrease in agricultural land area over time in the lower foothills. In the lowlands, there was a 9% (435 km^2) decrease in dryland farming over the whole period, and there was a similar reduction in overall agricultural land. However, orchards and vineyards increased by 52 km^2 and plantations decreased by 62 km^2, so there was not a one to one shift from dryland to fallow land. In the estuary sub-basin the overall decrease in total agricultural land use was driven largely by a decrease in the extent of dryland farming.

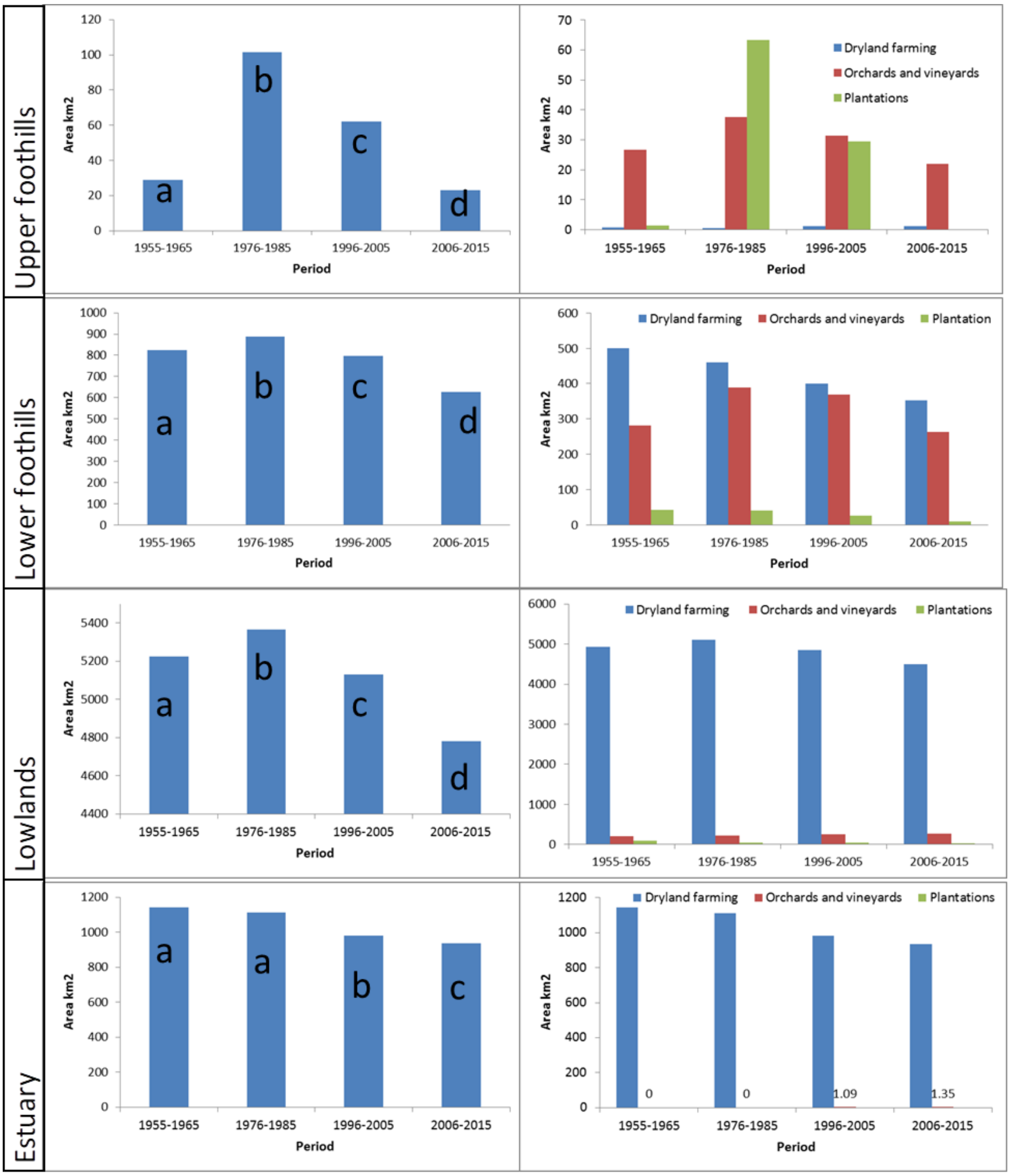

Figure 3.6 **Change in area of total agricultural land, dryland farming, orchards and vineyards, plantations and urban areas between time periods and separated into sub-basins. Letters differing between periods indicate significant differences (p<0.05) (e.g. top left a and b) and where they are the same there is no significant difference (e.g. top right a and a)**

The number of farm dams in the basin increased dramatically over time (Appendix Table 8 and Figure 3.7), with a gain of ±179 dams between 1955 and 2005, followed by a slight decrease of ±21 dams from 2005 to 2015. Between 1985 and 2005 the number of dams more than doubled in the upper and lowlands, and almost doubled in the lower foothills and estuary.

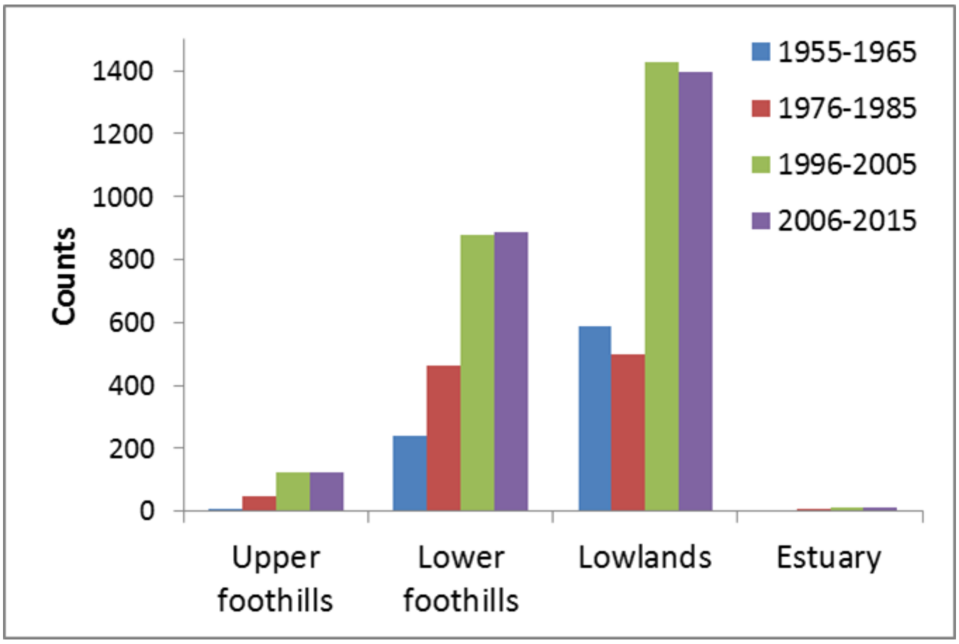

Figure 3.7 Total number of farm dams across the basin

3.4.2 Rates of change in land-use

Agricultural land increased in extent at a rate of +12.4 km^2 per year (1955-1965 to 1976-1985), thereafter decreased at an accelerated rate of -24.7 km^2 per year (1976-1985 to 1996-2005), to -40.5 km^2 per year (1996-2005 to 2006-2015) (Table 3.8). This was largely driven by a reduction in the rate of change of dryland farming and more recently due to reductions in the rates of change in orchards and vineyards and plantations.

The rates of change in extent of urban areas varied, but continually increased by +0.4 km^2 per year, +2.3 km^2 per year, and +0.8 km^2 per year respectively. Of these, towns and townships differed. The area of towns increased at rates of +0.2 km^2 to +1.8 km^2 then more slowly at a rate of +0.2 km^2 per year while that of townships accelerated progressively from +0.2 to +0.4 then to +0.6 km^2 per year over the same time periods (Table 3.8). The extent of undeveloped land initially decreased at a rate of -12.7 km^2 per year, after which it increased by +22.3 km^2 and +39.69 km^2 per year (Table 3.8).

The rates of change in numbers of farms decreased over time while that of townships increased. There were no noticeable differences in the rates of change of the number of industrial buildings and towns over time. Waterbodies increased at an average rate of +8.2 per year and by +79.3 per year, and then decreased by -1.6 per year. Of these, dams increased by +8.9 followed by a large increase of +71.2 per year after which the rate dropped and decreased by -1.4 dams per year (Table 3.8).

Table 3.8 Rates of change in land-use between the study periods

Land-use classes	Rate of change between periods (km²/year)		
	1955-1965 to 1976-1985	1976-1985 to 1996-2005	1996-2005 to 2006-2015
Dryland farming	4.85	-22.20	-29.55
Orchards and vineyards	6.64	-0.13	-6.5
Plantation	0.87	-2.36	-4.34
Total agricultural	**12.36**	**-24.69**	**-40.46**
Towns	0.15	1.82	0.19
Townships	0.20	0.48	0.57
Urban area total	0.36	2.31	0.76
Undeveloped land	-12.72	22.38	39.69
Infrastructure types	Rate of change between periods (Counts/year)		
Farms	-9.60	-0.55	-20.2
Industrial buildings	-0.05	-0.05	0.46
Towns	0.15	0	0.66
Townships	-0.90	5.40	16.73
Total buildings	**-10.40**	**4.80**	**-2.33**
Dams	8.95	71.20	-1.4
Dry pans	-3.85	-2.15	0
Non-perennial pans	1.70	10	-0.2
Perennial pans	1.45	0.30	-0.06
Total water bodies	**8.25**	**79.35**	**-1.66**

3.5 Discussion

Contrary to expectations, changes in land-use were not progressive throughout the study period. Conversion of natural vegetation and/or cultivated area to urban was the only progressive change in land-use; and urban areas increased by ± 1%. This relatively small increase in the coverage of urban areas in the catchment belies two important factors: (1) the (negative) influence that in-basin urban areas tend to exert on downstream ecosystems, mainly in terms of water usage and water pollution, and; (2) the negative influence of increased urban areas outside of the basin, for instance greater Cape Town and Stellenbosch areas, which draw some of their water supplies from the basin's rivers. Tizora *et al.* (2016) reported that in the past 24 years, increases in urban areas were mostly concentrated within the Cape Metropolitan area and the Cape Winelands district.

The other major land-uses in the basin, cultivation and plantations, declined by ±10% over the study period; and undeveloped land, which includes land recovering from cultivation or plantations, increased by almost the same amount (8%). Similar trends in land-use were reported by Stuckenberg (2013) for the Berg River Basin from 1986/1987 to 2007, based on an analysis of aerial satellite imagery using remote sensing techniques. Comparable findings were reported for the Cape Metropolitan area, with an 8.49% reduction in cultivation between 1990 and 2014 (Tizora *et al.* 2016). The increase in undeveloped land is fairly recent, with most clearing of woody vegetation having taken place since the start of the Working for Water programme, which is tasked with clearing exotic woody forests to reduce water lost via transpiration (Albhaisi *et al.* 2013). A subsequent goal of the programme is to improve the ecological condition of rivers and the lotic landscape overall, with the expectation that this will also increase flow in rivers to buffer that required by the Ecological Reserve as required by National Water Act 36 of 1998 (DWAF 1998). Losses of plantations were also due to forest fires near Franschhoek (Garcia-Ruiz 2003; Currie *et al.* 2009; Albhaisi *et al.* 2013). A general decrease of plantations in the Western Cape has been associated with fires and government's forestry exit policy; more than 44 000 hectares of forestry plantations were decommissioned in 2001 (Tizora *et al.* 2016). There have been numerous recent attempts to re-vegetate the lotic landscape in the Berg River Basin, but the outcome of these efforts were not necessarily captured in the maps that were available for analysis. In most cases plantation removal creates 'undeveloped land', which may or not recover either naturally (Currie *et al.* 2009) or with assistance. The bulk of this recovering land is in the headwaters, however, upstream of the newly constructed Berg River Dam, so any benefits that may have accrued to the river ecosystem from having a portion of its basin return to a more natural condition are offset by the barrier effect and flow regulation of the Berg River Dam.

The decline in cultivated area also does not imply an automatic improvement in ecological condition. For example, the replacement of agricultural lands by undeveloped land and urban areas showed that initially (1955-1965 to 1976-1985) the extent of agricultural land increased while that of undeveloped land decreased, thereafter the extent of each moved in the other direction, with agricultural land decreasing from ±7466 km^2 to ±6365 km^2 (1975-1985 to 2005-2015) while undeveloped land increased from ±1430 km^2 to ±2473 km^2 over the same time. Across South Africa the area of land used for the cultivation of maize, wheat and dairy has decreased significantly over the last 20 years (Agricultural Statistics 2008). This trend has also been reported for the Berg River Basin (Stuckenberg 2013; Albhaisi *et al.* 2013). This, in part, is also influenced by a reduction in the number of farms due to the reduced profitability of farming and general scarcity of water country wide. However, despite the decline in acreage under cultivation, production remains relatively constant, indicating a shifting trend towards intensified agriculture; changes in irrigation techniques, improved fertilizers, more efficient mechanisation and the use of drought- and pathogen-resistant genetically-modified seeds (du Plessis 2004). None of these trends bode particularly well for the river ecosystems of the valley. The decrease in cultivation may also be linked to alien infestation across the country, i.e. invasive alien species cost South Africans tens of billions of rands annually in lost agricultural productivity and resources spent on invasive species control (Le Maitre *et al.* 2004). Both regionally and nationally South Africa is already facing critical water shortages; more water is lost to invading woody plants most of which have no commercial value and do not contribute to the country's economy (Le Maitre *et al.* 2004). The increase in purchase of agricultural lands (farms) for alternative uses may have contributed to the decrease in area of land under cultivation/irrigation. More farms within the middle and lower reaches of the basin have been bought by lifestyle farmers who are more focused on satisfying the need of an acquired life style than agricultural production (Reed and Kleynahans 2009). For instance lifestyle farmers will not necessarily plant the same number of hectares that a commercial production farmer would grow.

The abstraction and regulation of water flows in the basin is also evidenced by the change in number of farm dams. Farm dams increased from <1000 in 1955-1965 to >2500 in 1996-

2005 with the bulk of the increase taking place in the lower foothills and lowland reaches of the river.

In conclusion, agricultural land in the Berg River catchment decreased in overall extent and plantations were removed from the upper foothills. While the increases in undeveloped land should be deemed to have had a positive influence on the functioning of the river ecosystems in the basin, they were offset by development in other areas, chiefly increased water abstraction and regulation and the construction of the Berg River Dam. The combined influence of these changes on hydrology, channel structure and faunal community structure are addressed in the subsequent chapters of this dissertation.

4 Historical changes in the flow regime of the Berg River

4.1 Introduction

The over-arching aim of this chapter is to identify changes in the Berg River flow regime over time and to identify, if and where possible, the cause(s) thereof. The two main hypotheses explored are:

(1) Land-use and water resource developments in the Berg River Basin have affected the volume and distribution of flows in the Berg River.

(2) The effects of land-use change on the flow regime can be isolated from changes due to large impoundments and water-resource schemes (e.g., the Theewaterskloof-Berg Scheme, the Berg River Dam, Voelvlei Dam and Misverstand Dam).

These questions have been addressed by evaluating the seasonal pattern and quantity of flow regime in the river over time, and where possible linking these to developments in the basin.

The flow regime is the pattern and timing of high and low flows in rivers, which are dictated in part by the characteristics of its catchment and the local climate (McMahon 1982; Poff *et al.* 1997; Ractcliffe 2009), and in part by developments in its basin. The flow regime is regarded as the driver of river character because, to a large extent, it determines the nature of the river channel, sediments, water quality and the life these support (Poff and Ward 1989; Poff *et al.* 1997; Arthington 2002; Figure 4.1).

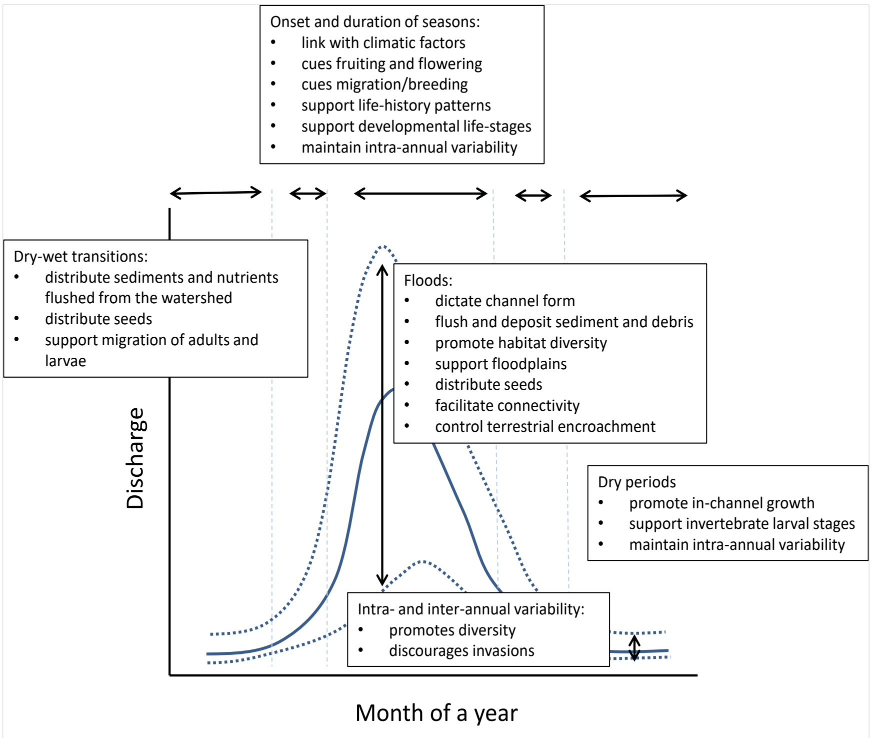

Figure 4.1 The importance of different parts of the flow regime (after Poff *et al.* 1997; Bunn and Arthington 2002)

The different "parts" of the flow regime, including its variability, together contribute to the overall maintenance of the river and all can be considered important (King 2003). Generally, flow within a year is divided into low and high flow seasons as well as transitional periods while, in more erratic systems there may be low flow periods interspersed with unpredictable flood events (Poff and Ward 1989). Low and high flows at different intervals, combined with other aspects of flow regime dynamics, help to define the habitat and thus aquatic species distributions and community composition (Allan 1995), and maintain connectivity of the whole ecosystem from source to mouth (Poff *et al.* 1997; Bunn and Arthington 2002; King *et al.* 2003a). They act as stresses or opportunities for aquatic species (Poff and Ward 1989) and the onset, duration and magnitude of flow in each flow season provide cues for life-cycle stages (Montgomery *et al.* 1993; Mahoney and Rood 1998; Tockner *et al.* 2000; Lauer and Pyron 2016). Rising and falling water levels transport and deposit sediments, aquatic biota and vegetative propagules to downstream locations (Goodson *et al.* 2001), which plays a role in structuring communities and maintaining species richness in aquatic ecosystems (Merrit and Wohl 2002; Chambert 2006; Richardson *et al.* 2007).

Water resource developments and land-use changes affect rivers' flow regimes, water chemistry and sediment and temperature regimes and, as a knock-on effect, their fauna and flora (Poff *et al.* 1997; King and Brown 2000; Bunn and Arthington 2002; King *et al.* 2003a). These changes also affect the people living near to and/or dependent on the rivers. Changes to the flow regime may alter the habitat and disrupt these cues and cause declines in abundance, health, or species richness (French and Chambers 1996; Poff *et al.* 1997; Yarnell *et al.* 2010).

According to Bunn and Arthington (2002), the loss of wet-dry cycles often has drastic ecological impacts that favour exotic species and changes have been demonstrated to result in impairments to water quality, physical habitat availability and structure, and the aquatic biota of rivers (Ligon *et al.* 1995; Pringle *et al.* 2000; Stromberg *et al.* 2001; Bunn and Arthington 2002; Zimmerman *et al.* 2010). Increased dry season flows may result in prolonged inundation of important habitats such as, for example, riffles (Bogan 1993). Conversely, Brown *et al.* (2006) showed that the ecological value of water to maintain some dry season flow in perennial rivers is extremely high and that in the Western Cape, with its long hot, dry summers, flow cessation during the dry season is extremely damaging to the river as riffles, runs and (often) pools dry up, leaving no riverine habitat (Brown *et al*. 2006). For instance, in the Berg River, many of the fish, such as *Barbus andrewi* spawn in the summer months and require sufficient inflow of oxygenated water into pools and over cobble gravel areas for spawning to be successful (Southern Waters 1996). Similarly, a loss of low flows in the McKenzie River in Oregon, led to a loss in habitat for juvenile salmon and a reduced population size (Ligon *et al.* 1995). When pools are isolated due the reduction in the dry season's flows, predation may increase (Gasith and Resh 1999), and when riffles dry out flow-dependent macroinvertebrate taxa (such as filter feeders) are negatively affected, with knock-on effects throughout the food chain. Extended periods of lower flows without small and intermediate floods may also lead to a reduction in the river's efficiency to transport incoming sediment, causing build-up of sediment deposits in the active channel and the covering of the exposed bedrock to form sand sheets and alluvial bars (Van Coller 1997) and reducing biotic diversity (Johnson 1994; Williams 1996).

Thus, changes in the pattern of flows in the Berg River would be expected to have a knock-on effect on the channel structure (Chapter 5), the quality and availability of habitats for riparian vegetation communities and macroinvertebrate communities (Chapter 6).

4.2 Study area

Changes in land-use (discussed in Chapter 3) were assessed in four periods 1955 to 1965, 1976 to 1985, 1996-2005, and 2006-2015, which are referred to here as '**land-use periods**' (**or LU periods**). Currently, more than 71% (6365.45 km^2) of the area of the Berg River Basin is under agricultural use; about 28% is undeveloped and just more than 1% is urban (Chapter 3, Table 3.6). The mountain stream section of the Berg River, located upstream of the Berg River Dam is largely undeveloped, although it was previously afforested, mainly with exotic pine species, and there is some water abstraction in places. The upper foothills of the basin include the town of Franschhoek and the settlement at Robertsvlei, and are dominated by orchards and vineyards (11% of the upper foothill area). The lower foothills of the basin include the now-extensive urban areas of Paarl, Dal Josafat and Wellington, and the smaller settlement of Hermon. The lower foothills are characterised by dryland farming (19%), together with orchards and vineyards (14.5%) (Chapter 3, Table 3.6). The lowlands of the basin include the settlements of Gouda, Riebeek-West, Bridgetown, Mooreesburg, Hopefield and Piketberg. They are characterised by dryland farming (43%), with some orchards and vineyards (2.5%) (Chapter 3, Table 3.6).

Several water resource development schemes have also been implemented over the years that have altered the flows in the Berg River. Early water management schemes included engineering interventions at the river mouth and diversions and weirs for water supply. For example, as early as 1852, a transfer of water from the Witte River via a furrow to the Krom River was implemented ('Gawie se Water'), and the damming of Voelvlei first began some decades later (DWAF, 2004). From 1950 onwards various large-scale engineering works and dams were developed, which became part of the Western Cape Water Supply System (WCWSS; DWS 2014) (Table 4.1); these developments are likely to have significantly impacted flow in the Berg River. In the 1950s, Wemmershoek Dam (58 Mm^3) and the seasonal wetland storage scheme at Voelvlei, were built. The capacity of the off channel Voelvlei Dam was increased in 1971 (158 Mm^3) and a few years later Misverstand Dam was built to supply to farmers and various West Coast urban settlements. In the late 1970s the Theewaterskloof Dam and its associated transfer tunnels were constructed to link the Riviersonderend catchment with that of the Berg and the Eerste rivers. From November 1980, the Theewaterskloof-Berg River Water Scheme (in the Breede River Basin) started to supplement dry-season flows in the Berg River catchment (Snaddon and Davies 1998) and at times supplied as much as 27% of the Berg River's flows (www.fewlbnexus.uct.ac.za, accessed 17 January 2017). In 2007, the Berg River Dam was completed, and the system of transfers to and from the Berg and Breede basins changed again (127 Mm^3; Table 4.1). Together with other inter-basin transfers they now form a complex system of pumping to and from parts of the basin and to and from adjacent basins, referred to as the Western Cape Water Supply System. There are also hundreds of smaller impoundments or farm dams (as referred to in Chapter 3, e.g. Appendix Table 4 to Appendix Table 7), each of which has a small individual impact on flow in the Berg River, although their cumulative impacts could be substantial (e.g Mantel *et al.* 2010).

Table 4.1 Major water resource developments in the Berg River Basin

Impoundment	Date established	Capacity (Mm3)	Sub-basin
Theewaterskloof-Berg River Scheme	1979	n/a	Upper foothills
Berg River Dam	2007	127 000	Upper foothills
Wemmershoek Dam	1957	58 644	Lower foothills
Voelvlei	1953; 1971 increased	158 600	Lowlands
Misverstand	1977	6 400	Lowlands

4.3 Methods

4.3.1 Data collation

Hydrological and rainfall data were used in this chapter. Daily river flows were sourced from the Department of Water and Sanitation (DWS) and rainfall from the South African Weather Services (SAWS). Rainfall data were used in data preparation (patching of flow data, Section 4.3.2 - Method 3) and in some of the "double mass plot" analyses to determine if and when changes occurred in rainfall's relationship with flow.

'Natural flows' refers to river flows in the absence of any human interventions. Because there are few, if any, records that predate human interventions, these data are usually modelled by adding back known abstractions from the river, and are called 'naturalised flows'. 'Historical flows' refer to the flows recorded in the river over time, which include the effect of successive human interventions.

4.3.1.1 Flow data collated

The DWS flow gauging weirs on the Berg River are listed in Table 4.2. The flow records were obtained for all of the listed stations, but several of these had fallen into disrepair, some had been discontinued and the records from gauges that are still in operation had numerous data gaps. Thus, only a subset of these gauges was used in subsequent analyses (Figure 4.2).

In Table 4.2, note that gauges G1H004, G1H076 and G1H077 form a set. G1H004 was discontinued in 2007 when it was flooded by the Berg River Dam. In 2008, it was replaced by G1H076 (upstream of the full supply level of the Berg River Dam) and G1H077 (downstream of the Berg River Dam). Data from G1H077 were thus added to those from G1H004 in order to analyse a full period from 1949 to 2016 (and referred to as G1H004-77). The extent of the gaps in each record are summarised in Table 4.3.

Table 4.2 DWS gauging weirs in the Berg River Basin

Gauge	Location	Upstream area (km^2)	Coordinates S	Coordinates E	Record start	Record end
			Mainstem			
G1H004	Bergriviershoek	70	33.92722	19.06083	1949-04-01	2007-05-17
G1H076	Upstream Berg River Dam	41	33.95578	19.07289	2008-03-14	2016-12-31
G1H077	Downstream Berg River Dam	83	33.90494	19.05478	2008-05-28	2016-08-31
G1H020	Daljosafat	628	33.70778	18.99111	1966-03-01	2016-12-31
G1H036	Vleesbank	1311	33.43500	18.95639	1979-01-01	2013-04-05
G1H013	Drie Heuwels	2936	33.13083	18.86278	1964-12-01	2016-12-31
G1H075	Misverstand Dam	3972	33.02383	18.78856	2006-09-03	2016-08-31
			Tributaries			
G1H003	Franschhoek River at Le Mouillage	47	33.89083	19.07889	1949-03-21	2014-10-23
G1H080	Wemmershoek River at the Dam	118	33.85240	19.04252	2009-06-26	2014-04-03
G1H078	Dwars River at Boschendal	55	33.87358	18.98214	2008-03-14	2014-10-23
G1H008	Little Berg River at Nieuwkloof	393	33.31389	19.07472	1954-05-01	2014-10-30
G1H034	Moorreesburg Spruit at Holle River	132	33.06667	18.75944	1976-07-20	2014-10-29
G1H039	Doring River at Grensplaas	44	33.53778	18.92111	1978-12-14	2014-10-28
G1H040	Fish River at La Fontaine	36	33.35639	18.95611	1979-08-16	2014-10-28
G1H041	Kompanjies River at De Eikeboomen	121	33.47972	18.97750	1979-08-30	2014-10-28
G1H043	Sand Spruit at Vrisgewaagd	151	33.16111	18.89222	1980-05-06	2014-10-29

Table 4.3 Gaps that were patched in hydrological records used in the analysis. List excludes gaps shorter than 15 days. Shaded rows indicate the gauges used in the bulk of this chapter

Gauge	Location	Length of record (years)	Number of gaps	Longest gap (days)	Patched for this study?
G1H004	Bergriviershoek	58.16	8	762	Yes
G1H076	Upstream Berg River Dam	6.61	0	0	Yes
G1H077	Downstream Berg River Dam	6.41	0	0	Yes
G1H020	Daljosafat	48.70	1	24	Yes
G1H036	Vleesbank	34.12	8	183	Yes
G1H013	Drie Heuwels	50.50	4	153	Yes
G1H075	Misverstand Dam	8.30	16	122	Yes
G1H003	Franschhoek River at Le Mouillage	65.64	21	731	No
G1H080	Wemmershoek River at Wemmershoek Dam	4.77	2	296	No
G1H078	Dwars River at Boschendal	6.61	1	17	No
G1H008	Little Berg River at Nieuwkloof	60.54	6	321	No
G1H034	Moorreesburg Spruit at Holle River	38.30	5	120	No
G1H039	Doring River at Grensplaas	35.90	1	64	No
G1H040	Fish River at La Fontaine	35.22	1	106	No
G1H041	Kompanjies River at De Eikeboomen	35.19	2	8	No
G1H043	Sand Spruit at Vrisgewaagd	34.50	1	30	No

Only the records from the gauges on the main Berg River (Table 4.3 and Table 4.3) were patched and analysed in detail. The records from gauging weirs in the tributaries could not be patched in a manner that was useful to the study, mainly because there were no reference stations that could be used for the patching. Data from gauges on some tributaries were, however, used in the patching process. The process adopted for patching the records is explained in Appendix B and summarised in Section 4.3.2.

4.3.1.2 Rainfall data collated

Rainfall data were acquired from the stations listed in Table 4.4 and used to patch flow data where necessary, and to compare relationships between flow and rainfall during different periods.

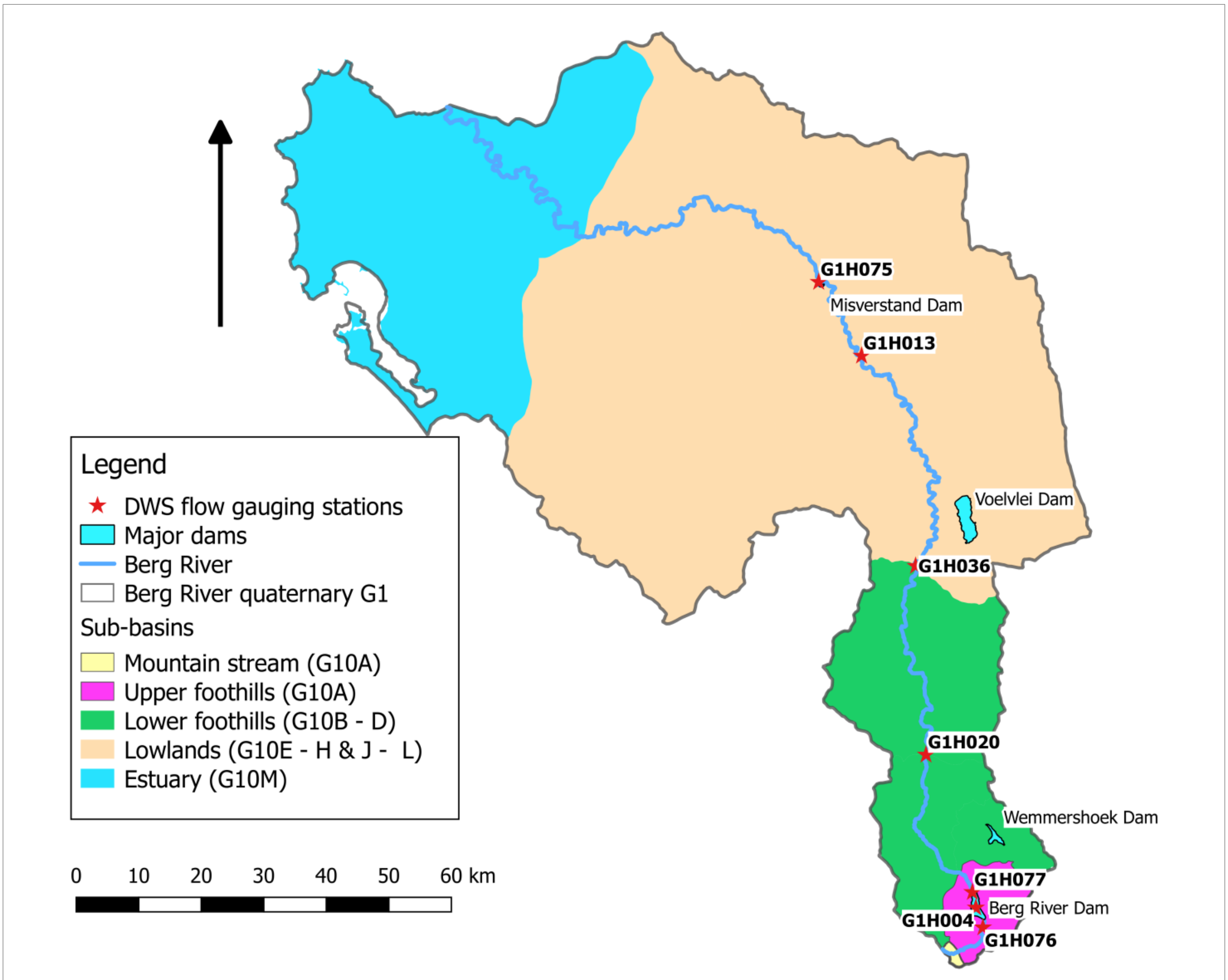

Figure 4.2 Location of DWA flow gauges on the main Berg River that were patched for use in the study

Table 4.4 Rainfall gauges used

Station number	Location and gauge number	Co-ordinates S	Co-ordinates E	Period	years
SAWS Paarl 1	Paarl (00 21823 0)	-33.721	18.972	1900-2015	115
SAWS Paarl 2	Paarl-TNK (00 21824 2)	-33.733	18.967	1978-1993	15
SAWS Paarl 3	Paarl- (00 21825 4)	-33.75	18.967	1938-1998	60

4.3.2 Patching of hydrological records

Three methods were used for patching the daily hydrological records, depending on the size of the gap and the availability of a reference flow gauge from which to estimate monthly volumes and distributions of daily flow. The flow and rain gauges that were used for patching are given in Table 4.5. In brief, the three methods were used:

1. Method 1: (for data gaps of less than a month; reference flow gauge available). The volume and distribution of flow in the gauge with missing data were estimated by comparison with those from the gauge used for patching, based on the relative sizes of their MARs (Mean Annual Runoff).
2. Method 2: (for data gaps of more than a month; reference flow gauge available). The missing volume was estimated using regressions developed for the gauge with missing data, and either another flow gauge, or a rainfall gauge (or an average of the two estimates).
3. Method 3: (for data gaps of more than a month; reference flow gauge not available). The missing volume was estimated using a regression relationship with rainfall, and apportioned using an “average” distribution of flow for wet or dry months for that flow gauge.

The type of gap (a couple of days to months) determined the method of patching to be used; therefore more than one method could be used on one gauge. Details of the three methods of patching, together with worked examples of each are provided in Appendix B. In addition, an example of how gauge G1H004 was patched using different flow gauges and rainfall for all three methods is given below.

Table 4.5 Gauges used for patching missing data

Zone	Gauge with missing data	Gauges used for patching
Upper foothills	G1H004 and G1H004-77	G1H003, G1H020 and SAWS Paarl 1
Lower foothills	G1H020	G1H004, G1H036 and SAWS Paarl 1
	G1H036	G1H020, G1H004 and SAWS Paarl 1
Lowlands	G1H013	G1H003, G1H020 and SAWS Paarl 1
	G1H075	G1H013

For example for G1H004 and G1H004-77, missing data were patched using the flow gauges G1H003 and G1H020, and rainfall data from SAWS Paarl 1 (Figure 4.3 to Figure 4.5). G1H003 was used to patch most gaps for the earlier period (up to March 1966), based on the relationships in Figure 4.3, and thereafter G1H020 was used (based on the relationships in Figure 4.4).

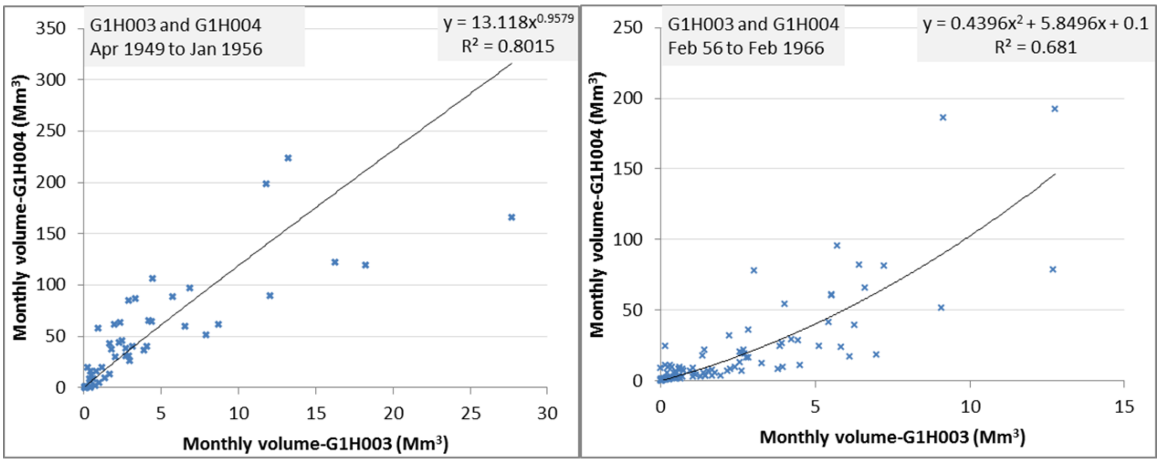

Figure 4.3 The relationship between flow at G1H004 and G1H003 for the period April 1949 to January 1956 (left). For the period February 1956 to February 1966 (right)

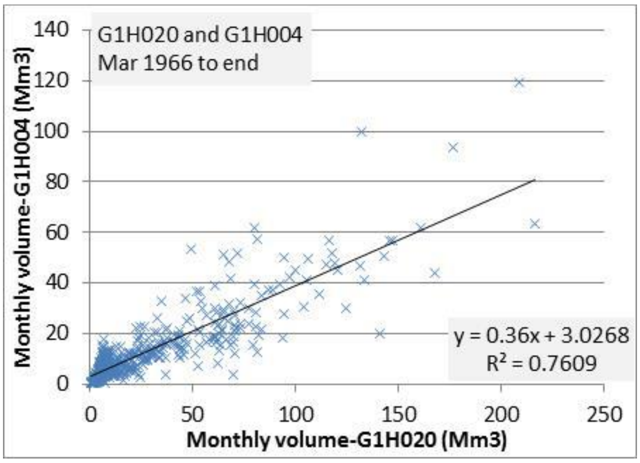

Figure 4.4 The relationship between flow at G1H004 and G1H020 for the period March 1966 to the end of G1H004's flow record (30 April 2007)

The relationship between flow and rainfall data derived using the data from April 1949 to May 1955 is shown in Figure 4.5 (left). The relationship for the longer period (April 1949 to December 1959) had a much lower R^2 and is shown in Figure 4.5 (right). After 1959 the relationship was even weaker (not shown). Thus the extensive data gap in G1H004 data from 01/06/1952 to 31/05/1954 was patched using y = 0.5837 x + 0.5 (Figure 4.5, left; Appendix B). The daily volume for the period from 1951 to 1954, which includes the patched gap is shown in Figure 4.6.

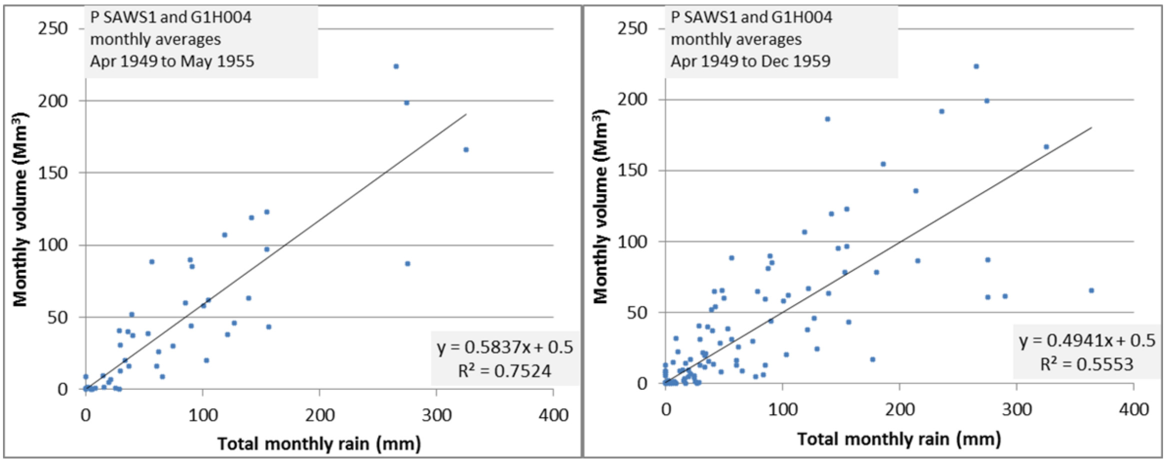

Figure 4.5 **Relationship between rainfall and flow at G1H004 for patching Method 3 for the period 1949 to 1955 (left). The relationship extended to December 1959 (right)**

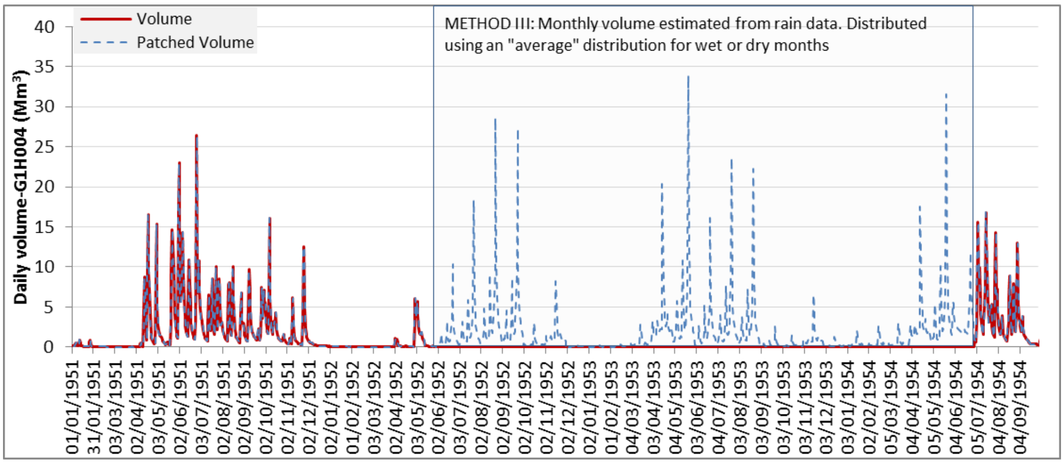

Figure 4.6 **Daily hydrograph for G1H004 showing an extended data gap from 01/06/1952 - 31/05/1954 which was patched using Method 3.**

4.3.3 Calculation of flow indicators

After data sets were patched, the daily flow time series were imported into the DRIFT software (www.drift-eflows.com; King *et al.* 2003a; Brown *et al.* 2013). In DRIFT, a set of ecologically-relevant flow indicators (Table 4.6) was calculated for each gauge according to four seasons: the dry season (d) the wet or flood season (w or f), the transition season between dry and wet seasons (T1), and the transition season after the wet season (T2). DRIFT flow indicators were generated for the Berg River mainstem gauges with more than 10 years of flow data: G1H004 (together with G1H077) in the upper foothills, G1H020 and G1H036 in the lower foothills, and G1H013 in the lowlands.

DRIFT characterises flow in terms of the:

- pattern of flow: the timing (onset) and duration of wet, dry and transitional seasons;

- magnitudes of flows: minimum dry season flow, maximum flood season flow, and average flows for each season;
- high flows: For "flashy" flow regimes, such as those characteristic of the Western Cape, higher flows or peak events are allocated to one of eight flood classes and frequencies calculated:
  - Four intra-annual floods (Class1, Class2, Class3, and Class4)
  - Four inter-annual floods with return periods of 2, 5, 10 and 20 years.

Table 4.6 The ecologically-relevant flow indicators generated using DRIFT software

Flow indicator	Code	Units
Mean annual runoff	MAR	$m^3 \times 10^6$ (or Mm^3) [1]
Dry season onset	Do	Calendar week
Dry season duration	Dd	Days
Dry season minimum flow (taken from 5 day running average)	Dq	m^3/s
Dry season average daily volume	Ddv	$m^3 \times 10^6$ (or Mm^3)
Flood season onset	Fo	Calendar week
Flood season peak (taken from 5-day running average)	Fq	m^3/s
Flood season average daily volume	Fdv	$m^3 \times 10^6$ (or Mm^3)
Flood season volume	Fv	$m^3 \times 10^6$ (or Mm^3)
Flood season duration (days)	Fd	Days
Number of intra-annual floods (Class 1, Class 2, Class 3, and Class 4) in the wet season[2]	C1w, C2w, C3w, C4w	Number per annum
Number of inter-annual floods with a return periods of 2, 5, 10 and 20 years	C5, C6, C7, C8	Number per annum

1. In some cases MAR is reported in m^3/s in this and other chapters
2. Note that the numbers of these floods are also calculated for the Dry, T1 and T2 seasons

The Berg River flow regime is "flashy" (James and King 2010), so flood events were separated from the daily average flow record using tools provided in the DRIFT DSS (Brown *et al.* 2013). In this DRIFT module, flood events are selected manually, by marking the beginning and end of each event. The software separates all the marked high flows from the low flows and then categorizes the floods according to eight size classes for inter- and intra-annual floods. In order to do this, the DRIFT software calculates the 1:2 year flood size (C5), and then calculates the size of the Class4 (C4w) flood (the largest intra-annual flood) by halving the 1:2 year flood (as halving the magnitude of an event results, in general terms, in a significant change in the sediment-moving power of the flood (Brown *et al.* 2013). The C3w flood size is calculated by halving the size of the C2w flood, and so on down to C1w.

DRIFT flow indicators were only analysed for the Berg River mainstem gauges with more than 10 years of flow data. These are: G1H004 (together with G1H077) in the upper foothills, G1H020 and G1H036 in the lower foothills, and G1H013 in the lowlands).

4.3.4 Data analyses

Flow data for gauges on the Berg River were analysed to identify the general characteristics of the flow regime, changes in flow regimes over time, and the possible reasons for these changes using regression equations, double-mass plots and tme-series graphs.

4.3.4.1 Regression equations

Regression equations were used to identify significant changes in the relationship between pairs of flow gauges and between flow and rain gauges, and 'best' equations found in terms of: (a) the Microsoft Excel relationship that yielded the highest R^2 value and (b) whichever data periods yielded the highest R^2. The regression analysis helped to identify gauges and periods which could be used for patching, and also to assess points at which these relationships changed. Regression equations were also used to identify significant trends in the flow indicators.

4.3.4.2 Double-mass plots

Double-mass plots were used to identify inflection points showing changes in the relationships between MAR and various DRIFT indicators, and between MAR and rainfall. A double-mass plot shows the cumulative values of one variable plotted against those of another. If the relationship between the two is constant the points will plot in a straight line (although small deviations above and below the line are expected). Changes in the slope (i.e. inflection points indicate a change in the relationship between the two variables for some reason (Searcy and Hardison 1960). In this study, inflections points could indicate: a change in land-use affecting flow at a particular flow gauge; a change in water resource developments affecting flow; that one or other gauge has (i) changed function; (ii) had an error.

Two types of double mass plot were used: (i) MAR versus rainfall, where inflections could be due to changes in land-use or water resource developments. For these rainfall at Paarl was used for all gauges because it had a long record and the pattern of rain was similar to the other rainfall stations (see Appendix Figure 2)., and; (ii) MAR against DRIFT indicators, where inflections could indicate changes in the relationship between MAR and other ecologically relevant characteristics of the flow regime. For these, the DRIFT indicators used were dry season onset (Do), dry season duration (Dd), dry season five-day average volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season average daily volume (Fdv) and two flood classes (Appendix B: Appendix Figure 3 to Appendix Figure 6).

Changes in slope were identified visually and checked by calculating differences in slope between successive years. Where the slope changed more than once in <3 years, the middle year was used. Only changes that occurred in both flow and rain, and in MAR-DRIFT plots were used in subsequent analyses. Once changes were identified, the averages of the variables for (i) the periods identified by the inflections points, (ii) the periods pre- and post- the main water resource developments, and (iii) the land-use periods used in Chapter 3 were assessed using T-tests to identify significant differences between periods. Given the number of data points per period (i.e. the number of years per period, which ranged from four to twelve), a lenient threshold was used for significance on the T-test (p-value ≤ 0.1).

4.3.4.3 Time-series graphs

Time-series graphs were created showing the years of change, the (three year) dry-season average flow, land-use (area of agricultural lands) and numbers of dams for the sub-basin where each gauge was located (e.g, for G1H004-77; Figure 4.17).

4.4 Results

4.4.1 Characteristics of flow regime

4.4.1.1 Hydrology of the Berg River

The naturalised MAR of the Berg River was estimated at between 908 Mm^3 (Ractliffe 2009) and 955 Mm^3 (DWS 2012; Aurecon 2016; Table 4.7); of this, 68% was from the upstream half of the basin and the Klein Berg River (Ractliffe 2009). In 2016, Aurecon (2016) estimated the historical MAR at ~488 Mm^3/a, which is about 46% lower than natural. The main reasons for the reduction are water abstractions for agriculture and major urban areas, including Franschhoek, Paarl, Wellington, Malmesbury, Riebeeck Wes, Riebeeck Kasteel, Piketberg, Porterville, Tulbagh, Saldana and the Greater Cape Town area (Gorgens and de Clercq 2005; Ractliffe *et al.* 2007; Leaner *et al.* 2012).

In Table 4.7, naturalised and historical data for G1H004 and G1H004-77 are sumarised with and without the period 1949 to 1959. The remaining analyses in this chapter omitted this period because recorded flows in this period were significantly, and probably erroneously, higher than other years, the data were also heavily patched (see Figure 4.7), and the relationship between flow and rainfall over this period differed from that immediately afterwards (Figure 4.5).

Table 4.7 Naturalised and historical MAR (Ractliffe 2009) and historical MAR from this study

Gauges	Ractliffe (2009)		This study	
	Naturalised MAR (Mm^3/annum)	Historical MAR for 1980-2004 (Mm^3/annum)	Historical MAR (patched data) (Mm^3/annum)	
G1H004	135	162	226[1] (1949-2007)	149 (1959-2007)
G1H004-77	n/a	n/a	209[1] (1949-2016)	142 (1959-2016)
G1H020	413	303	334 (1996-2016)	
G1H036	521	391	339 (1979-2013)	
G1H013	817	572	548 (1964-2016)	
G1H075	906	604	579 (2006-2016)	

4.4.1.2 Summary statistics

The annual, monthly and daily flow historic hydrographs and other summary statistics for the patched records at G1H004-77, G1H020, G1H036 and G1H013 are shown in Figure 4.7. Gauges G1H076 and G1H075 were excluded as their data records were too short for robust analysis. The data from all gauges resembled a typical Western Cape hydrograph (Nitsche

2000), although G1H036 had a relatively delayed peak, with highest flows in August instead of June/July. Monthly flows receded from October until March to April when they started rising again. The flows continued rising until they reached a maximum in July to August and started to decline again in September. The years with highest and lowest annual flows varied between the gauges, at G1H004-77, the highest flows were in 1977 (> 10 Mm^3) and the lowest in 2015 and 2016. At G1H020, the highest were in 1977 but also in 2013 (>22 Mm^3), and the lowest were in 2011 and 1972. At G1H036 and G1H013, highest annual flows occurred in 1991, 1983 and 2001 (>45 Mm^3 for both) and lowest in 2003 and 2011.

Figure 4.7 DRIFT summary statistics for on the Berg River. Note: There is no difference between the red and blue in the bars in the first graphs

4.4.1.3 Trends over time

Trends in the annual DRIFT flow indicators calculated using the data from G1H004-77, G1H020, G1H036 and G1H013 are shown in Figure 4.8, Figure 4.10, Figure 4.12, Figure 4.13 and Figure 4.14, respectively. For the within-year floods, the total number of floods in the year was used, rather than number of floods per season.

In the wet season, the picture is more subtle. In the upper catchment (G1H004-77), the wet season flows are lower, the wet season is longer, and the number of floods of all size classes is reduced over the ~40 years of record. The picture is similar at the next gauge downstream (G1H020), although the number of C5w floods increase. At the next gauge downstream, the reduction in size of the wet season flows and number of C3w floods is marked, and is accompanied by a shorter wet season (Fd), while the number of C4w floods increases. However, at the most downstream gauge (G1H013) some wet season indicators increase (slightly and not significantly). The total number of C1w to C5w decreases at all sites, although this is statistically significant only at the upper two sites. There are more significant changes at G1H004-77 and G1H036. The general trend is towards increased dry season flows and decreased wet season flows. In the dry season, all four gauges show increased dry season discharge, while in the lower two gauges (G1H036 and G1H013) this is accompanied by an increase in the duration of the dry season.

In the upper foothills, the data (G1H004-77) showed a slight, significant increase in Dq ($p<0.05$) and the duration of C2w and C3w floods; and significant decreases in Fq ($p<0.1$), Fdv ($p<0.1$), the number of C3w ($p<0.05$), C4w ($p<0.05$), and C5w ($p<0.05$) floods, the duration of and the total number of C1w to C5w floods (Figure 4.8). In the lower foothills at G1H020, the picture was less clear, and there was a significant increase in Dq ($p<0.05$), and a significant decrease in the number of C3w ($p<0.05$; Figure 4.10). There was also a significant decrease the total number of C1w to C5w floods. There were no significant changes in durations of flood events over time (Figure 4.11).

In the lower foothills at G1H036, Dd ($p<0.05$), MAR ($p<0.05$), Fq ($p<0.05$), Fdv ($p<0.05$), C3w ($p<0.05$), and C4w ($p<0.05$) decline significantly (Figure 4.12), and there were significant increases in the durations of C1w and C2w floods (Figure 4.13). In the lowlands (G1H013), there was a significant increase in Dd ($p<0.05$), and significant decreases in C2w ($p<0.05$), and a significant increase in C4w ($p<0.05$). The trends in floods may be because the C2w floods have become more “peaky”, and so moved into C3w (causing a decrease in C2w), and similarly with the C3w floods, these thereby becoming C4w (causing an increase). All floods showed significant decreases in the durations of each flood event (Figure 4.13). In addition, C3w showed a signifant increase in the size of the peak (not shown).

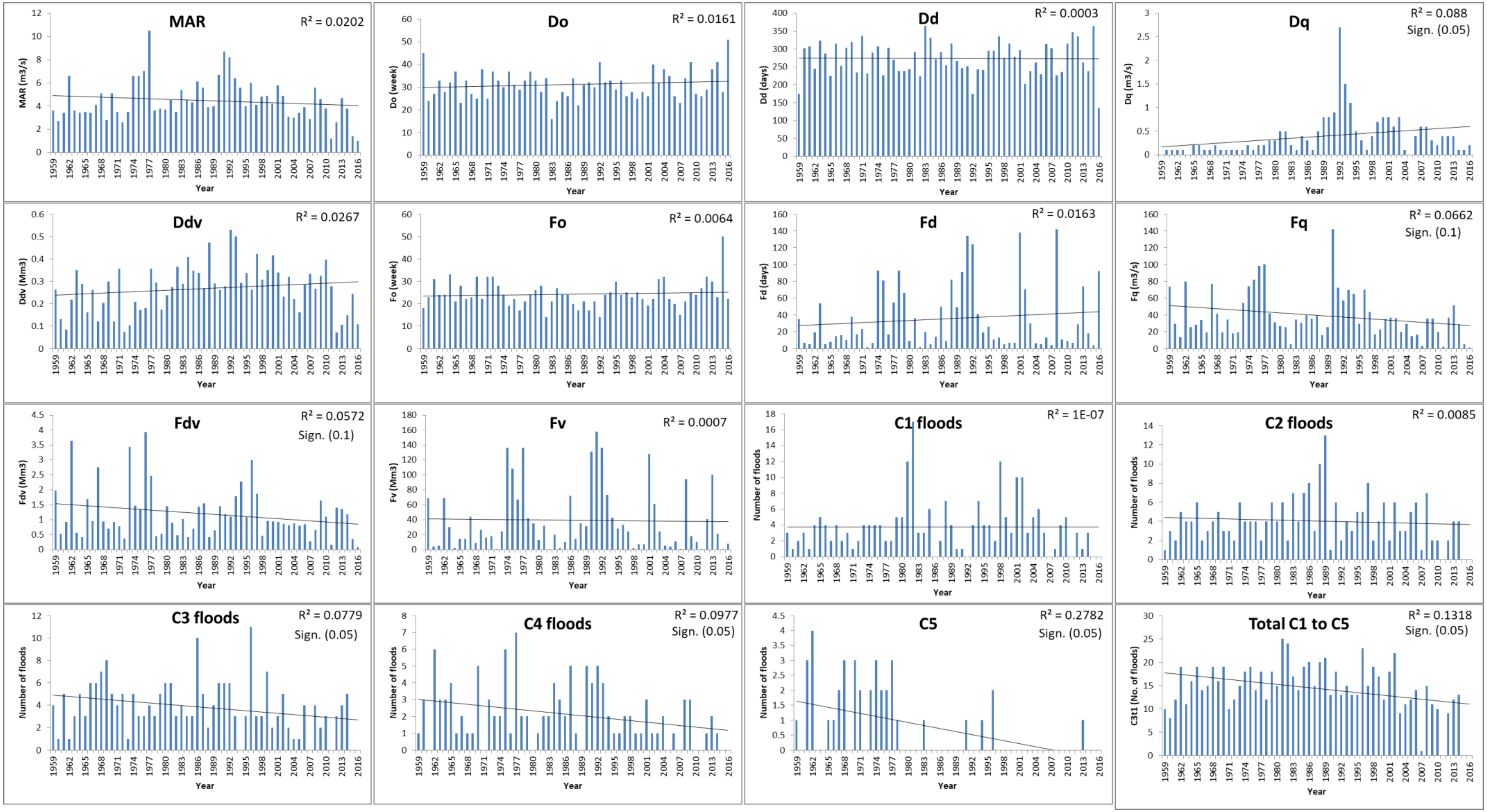

Figure 4.8 Trends in the DRIFT flow indicators calculated from the data for G1H004-77 at Bergriviershoek

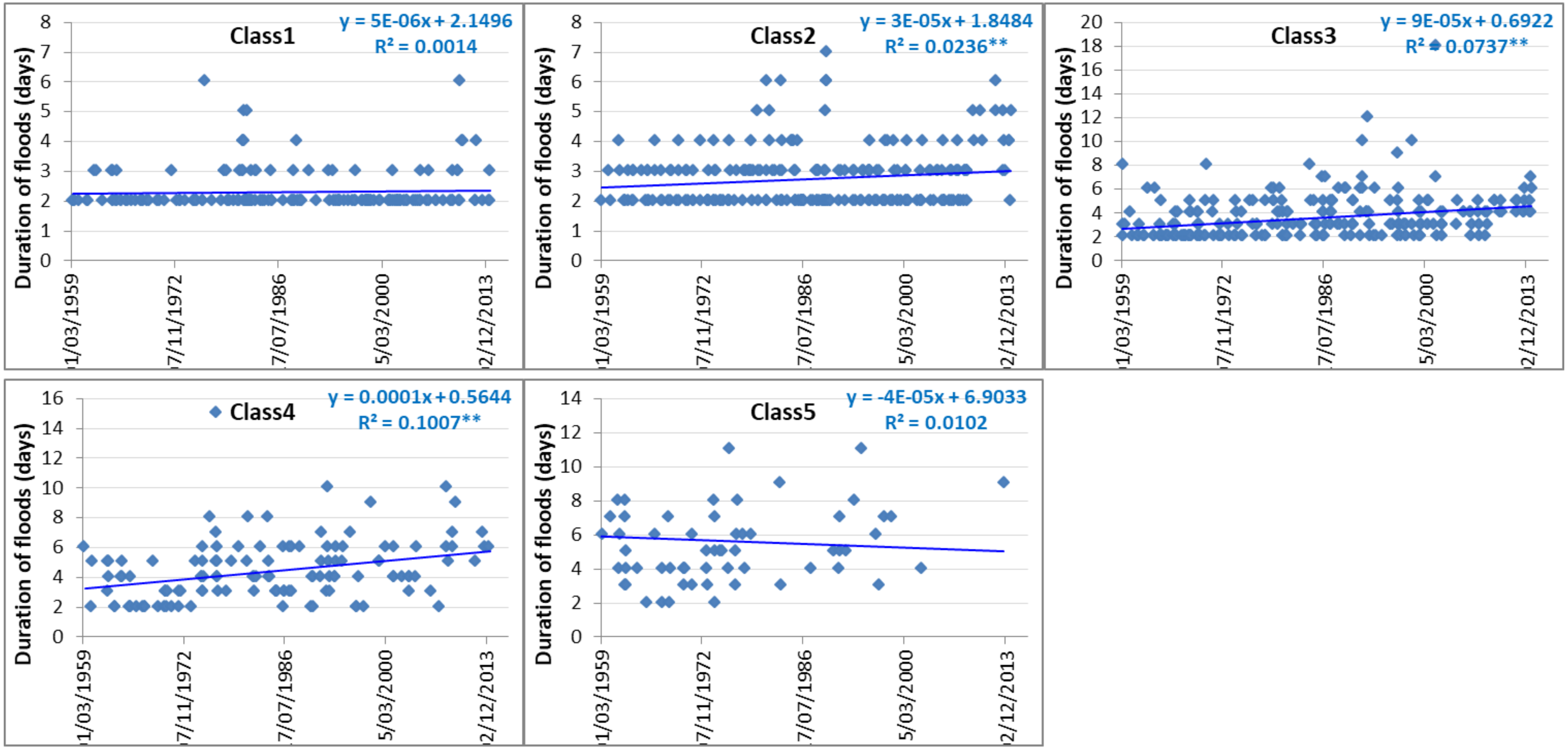

Figure 4.9 Trends in the duration of C1 to C5 floods over time (G1H004-77 at Bergrivershoek (: p< = 0.05, *, p< = 0.1)**

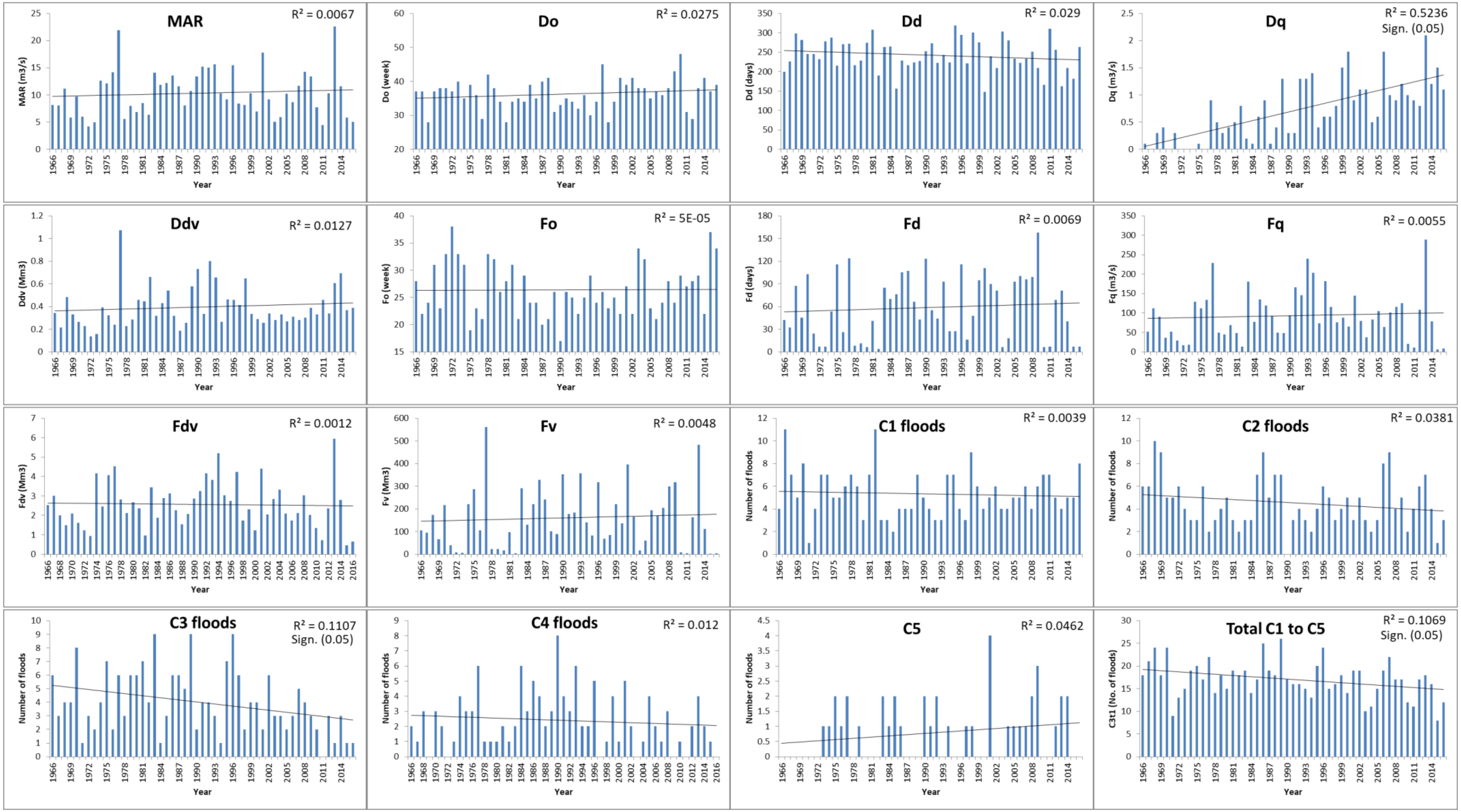

Figure 4.10 Trends in the DRIFT flow indicators calculated from the data from G1H020 at Daljosafat

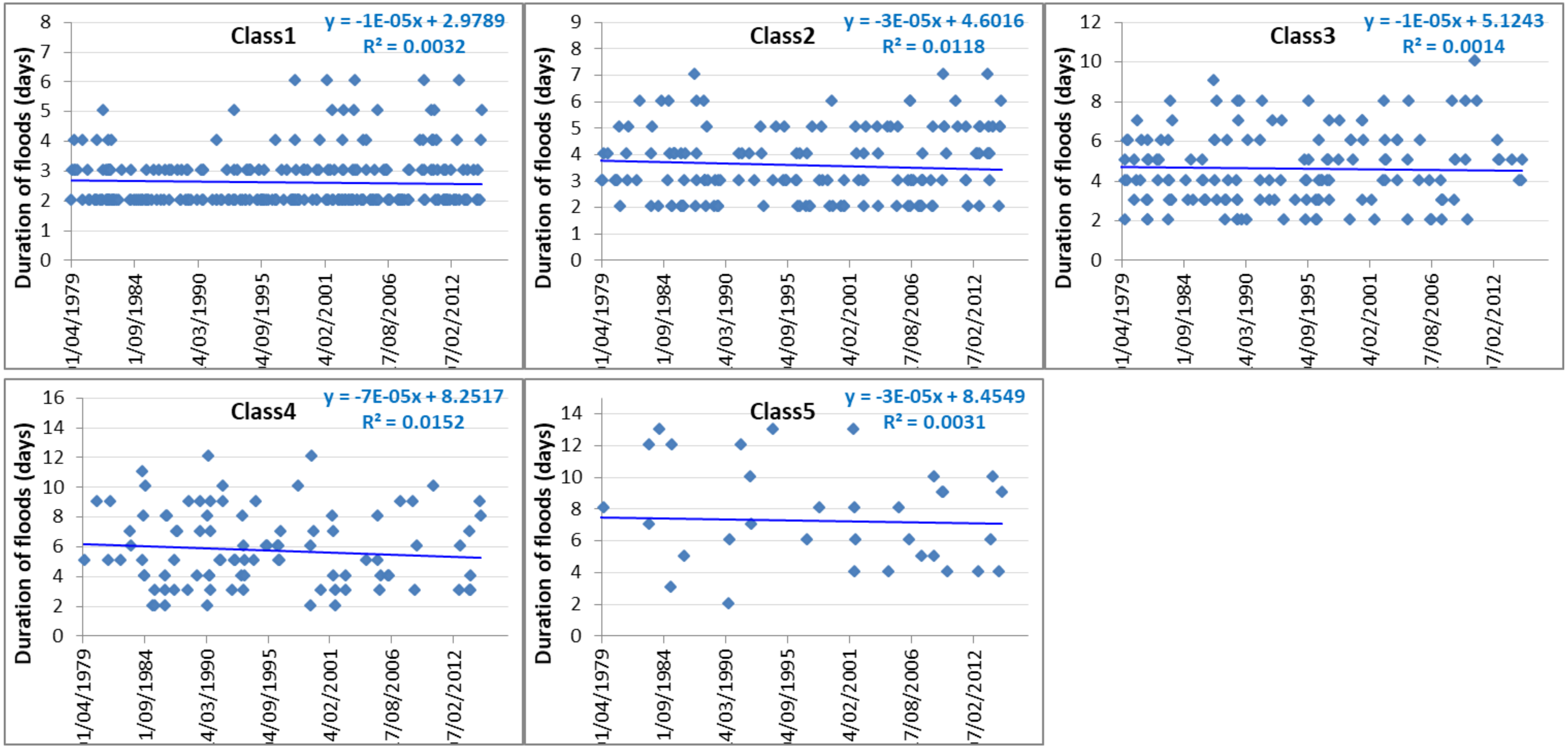

Figure 4.11 Trends in the duration of C1 to C5 floods over time (G1H020 at Daljosafat) (: p< = 0.05, *, p< = 0.1)**

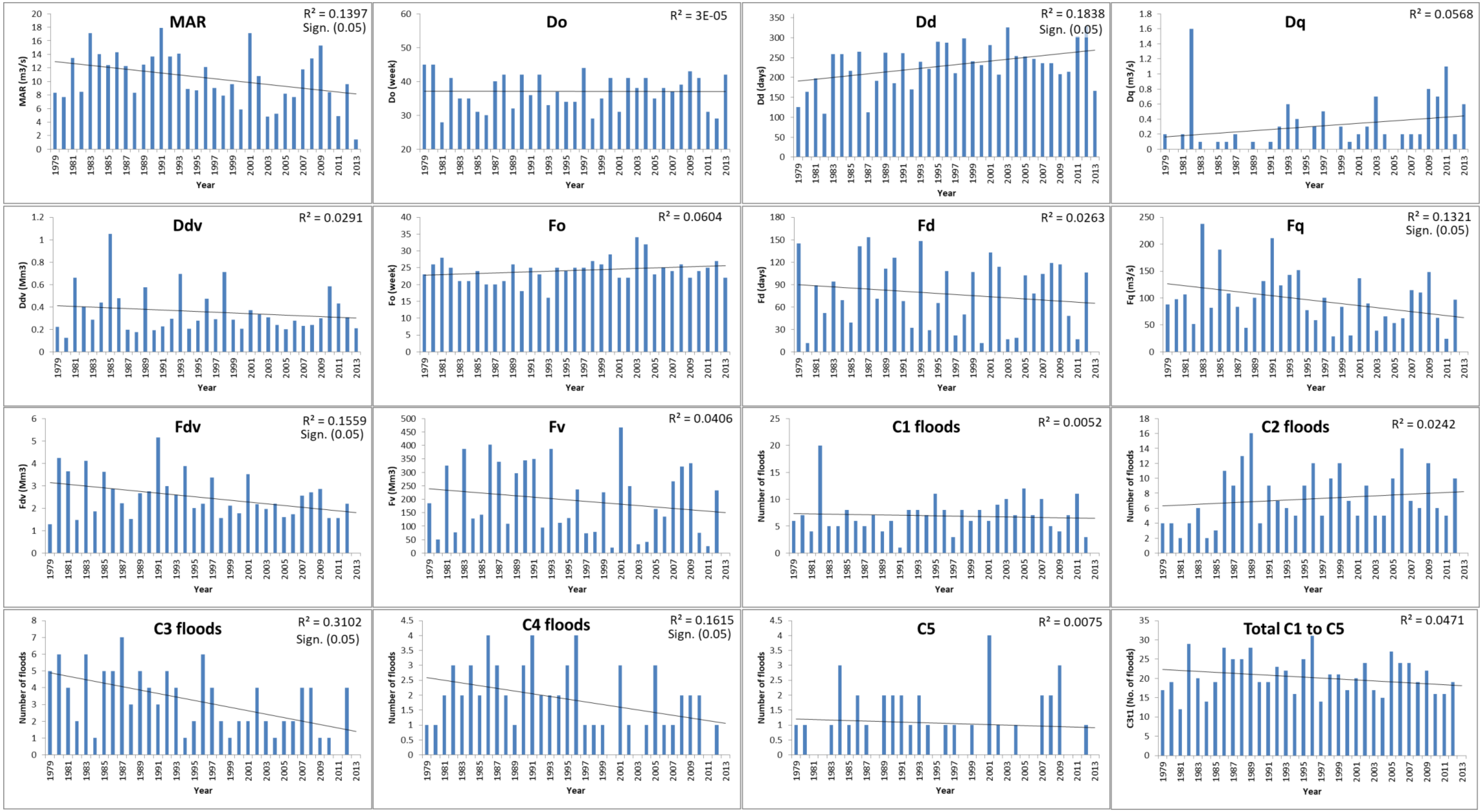

Figure 4.12 Trends in the DRIFT flow indicators calculated from the data from G1H036 at Vleesbank

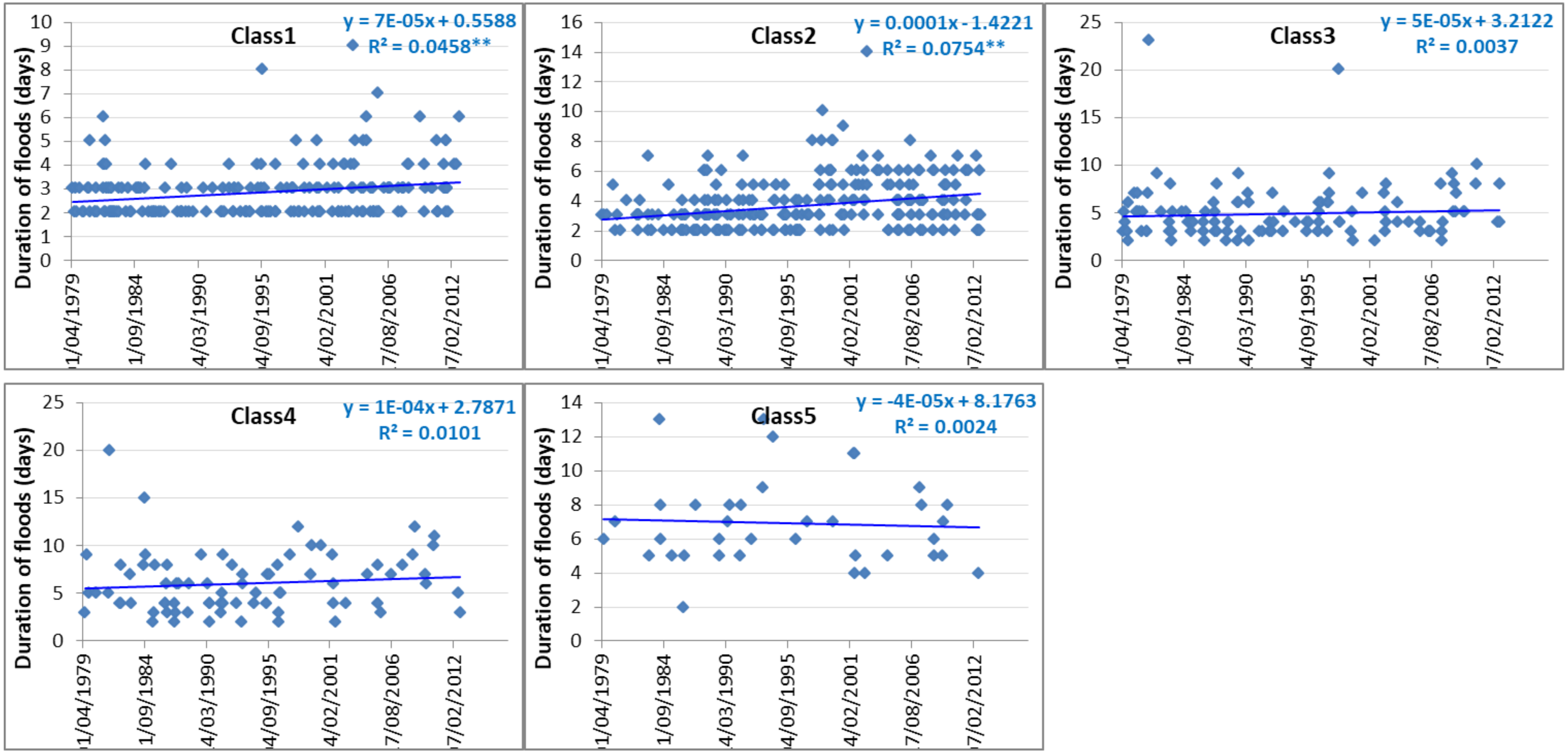

Figure 4.13 **Trends in the duration of C1 to C5 floods over time (G1H036 at Vleesbank) (**: p< = 0.05, *, p< = 0.1)**

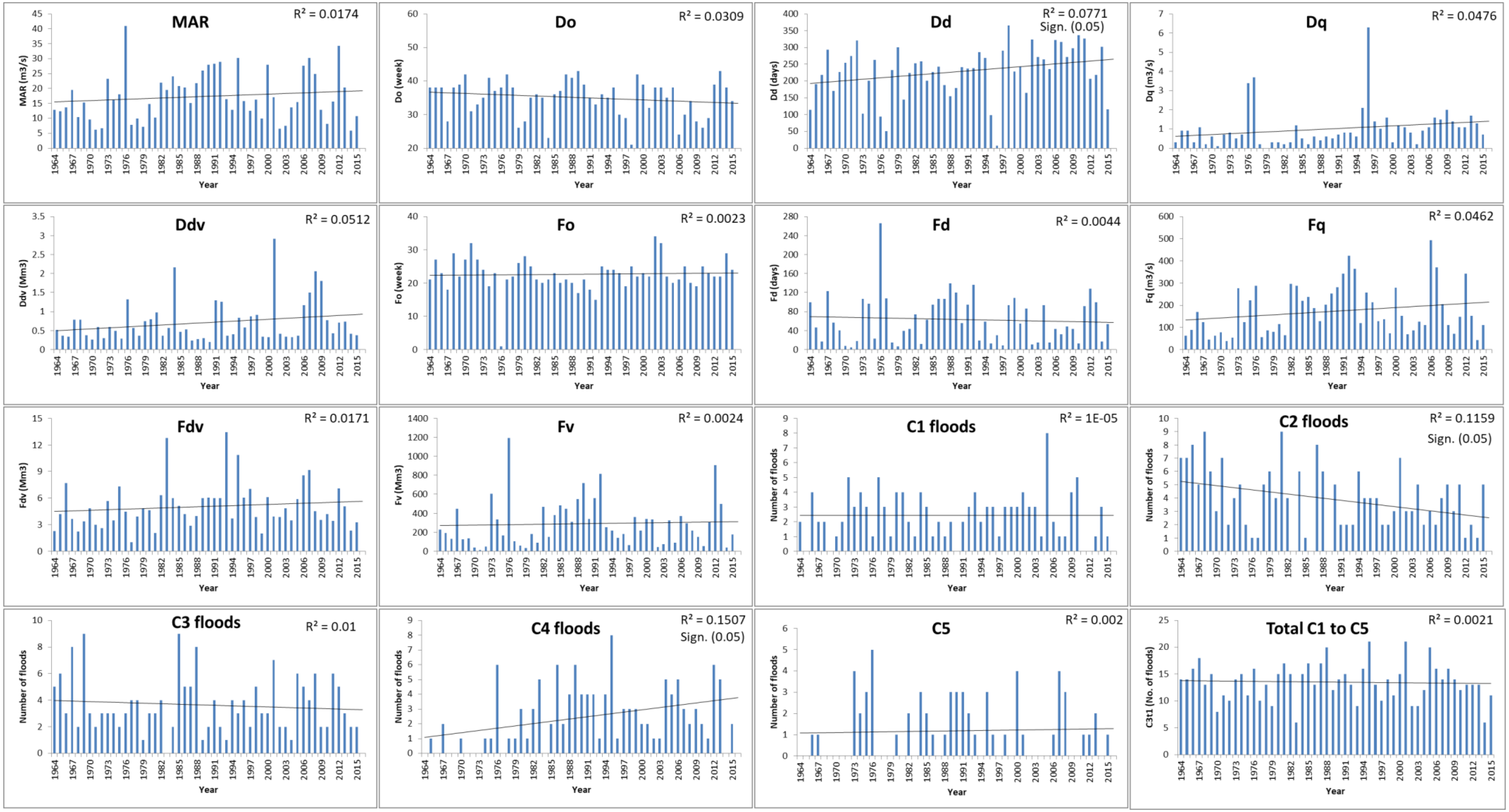

Figure 4.14 Trends in the DRIFT flow indicators calculated from the data from G1H013 at Drie Heuwels

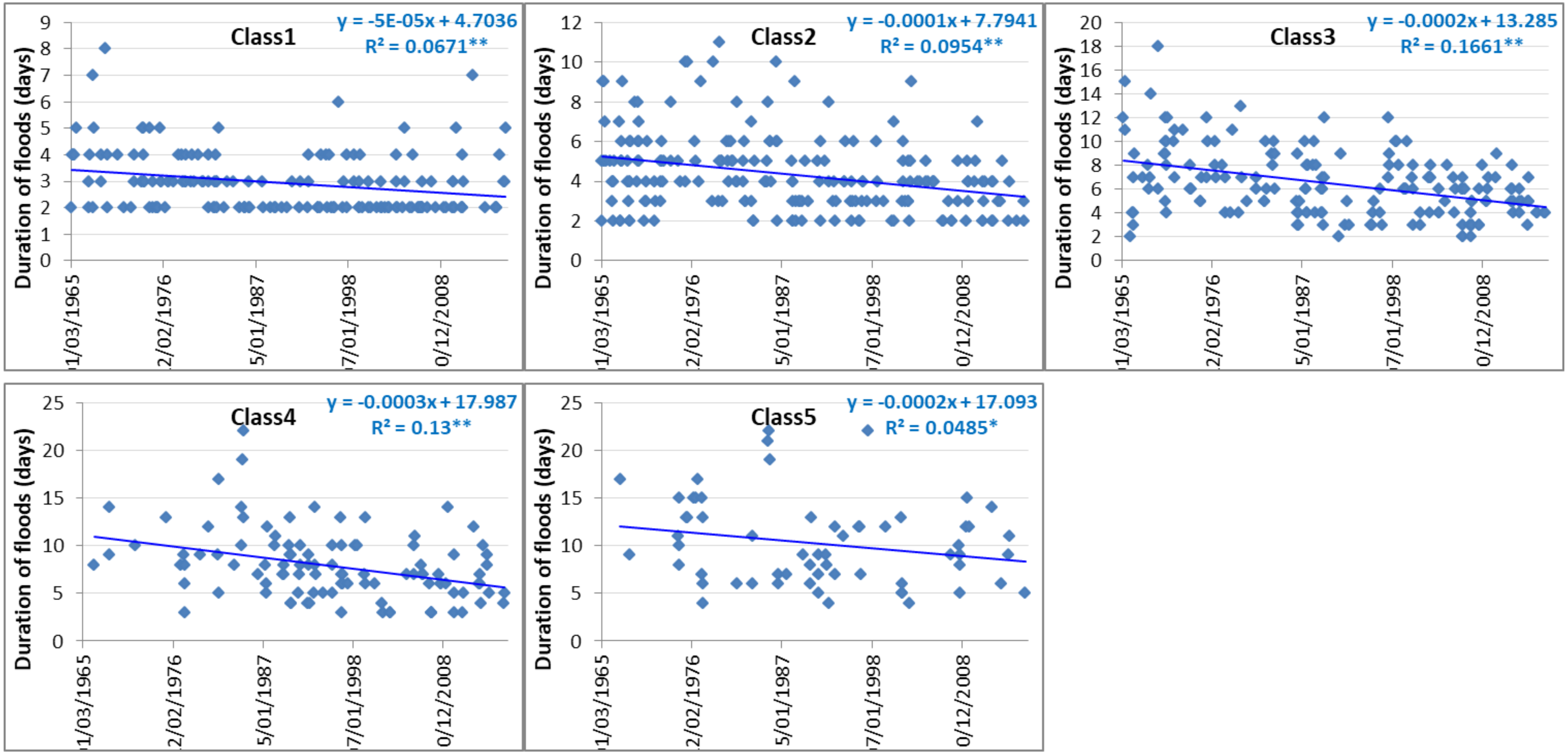

Figure 4.15 Trends in the duration of C1 to C5 floods over time (G1H013 at Drie Heuwels) (: p< = 0.05, *, p< = 0.1)**

4.4.2 Changes at particular times: Inflection points

4.4.2.1 Upper foothills sub-basin (G1H004-77 at Bergrivershoek)

Data from G1H004 were plotted for the period 1949 to 2007. The double-mass plots for the six main DRIFT indicators are shown in Figure 4.16, together with those for C3w (number of wet season class 3 floods), and the MAR-rainfall graph. The inflection points and dates are given on each graph. Double-mass plots were also constructed for the combined data from G1H004 and G1H0077 for the period 1988 to 2016 (Appendix Figure 3).

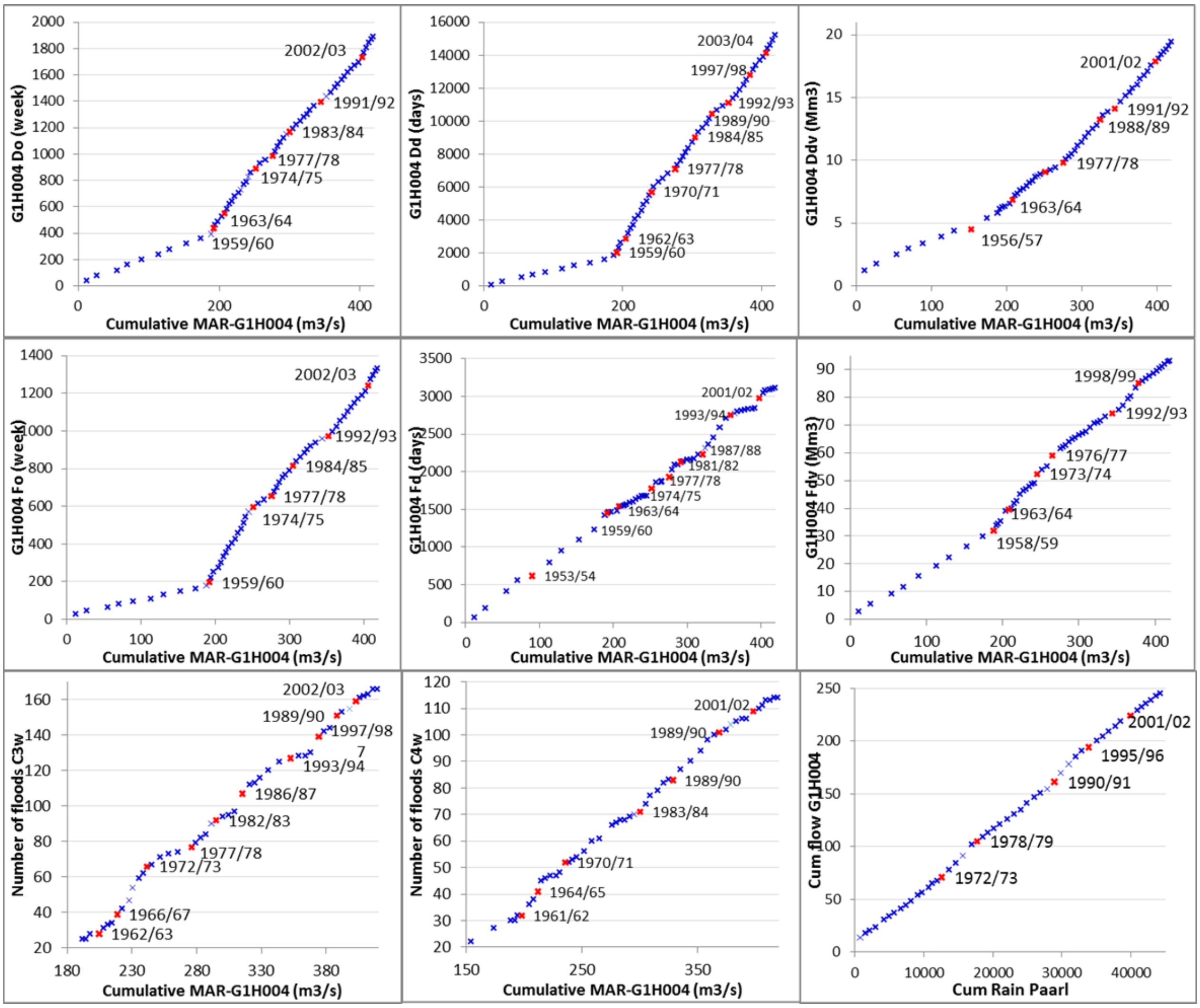

Figure 4.16 Double-mass plots of G1H004 MAR against dry season onset (Do), dry season duration (Dd), dry season volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season volume (Fdv), the number of Class 3 (C3w) and Class 4 (C4w) and cumulative rainfall

The inflection points in the MAR-DRIFT indicator relationships (taller vertical markers on Figure 4.17) coincide with those for the MAR-rain relationship (shorter vertical markers on Figure 4.17) for 1959/60, 1972/73, 1978/79, 1989/90, 1995/96, 2001/02, 2011/12. Figure 4.17 also shows: the three-year average dry season volume, the area under agriculture in the upper basin, and the number of dams upstream of the gauge to give some idea of the underlying changes that occurred.

The periods where inflection points occurred in both flow and rain and in MAR-DRIFT plots were 1959-1971, 1972-1978, 1979-1989, 1990-1995, 1996-2002, 2003-2011, 2012-2016. These are very close to the land-use periods analysed in Chapter 3 (Table 4.8).

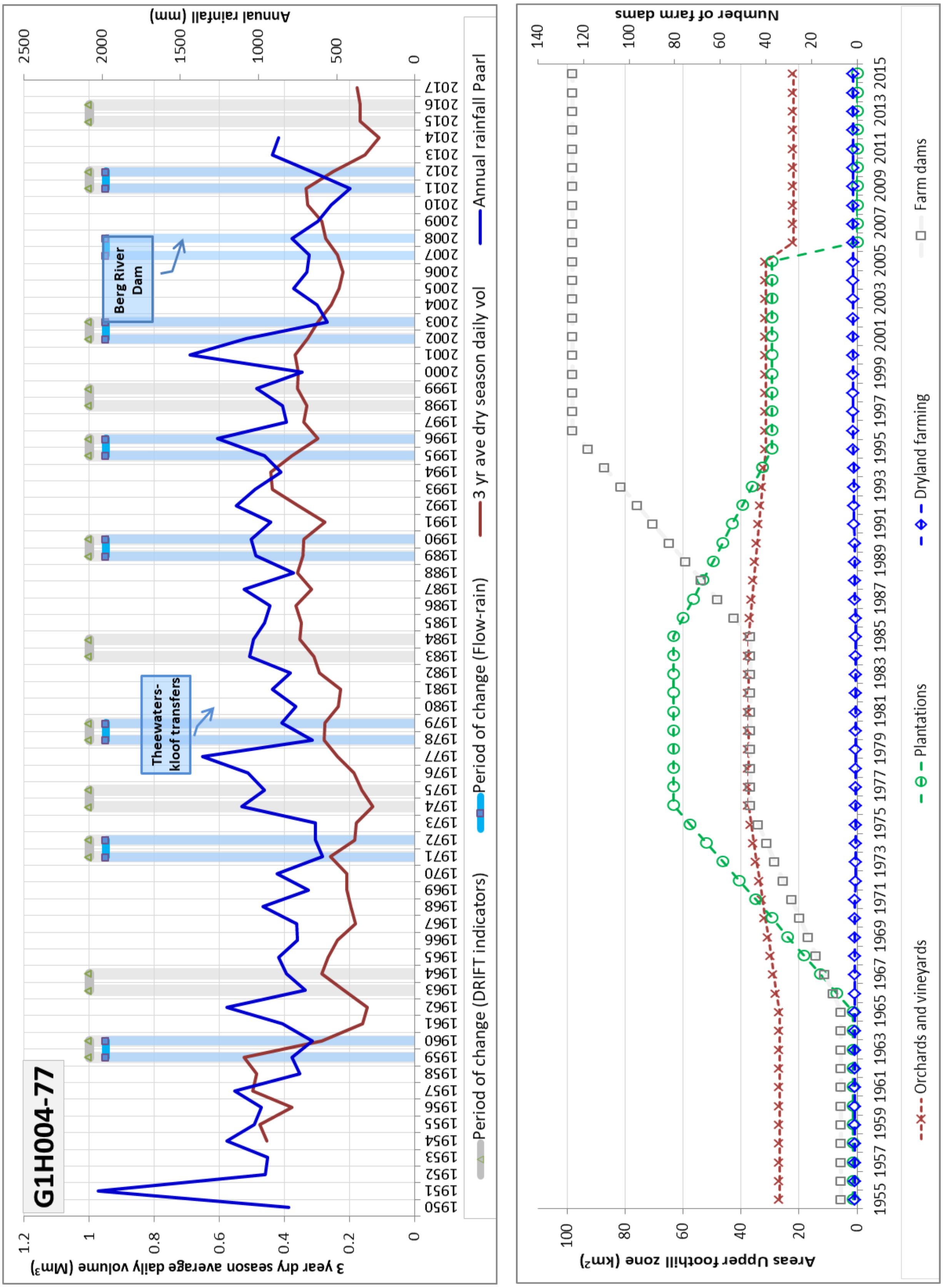

Figure 4.17 **(Left) Periods of change as determined by MAR-DRIFT indicators and the flow-rain relationship, 3-year average annual volume at G1H004-77, rainfall; (Right) Area of agricultural land-use, and number of dams in the upper foothills**

Table 4.8 G1H004-77 data: Periods where inflection points occurred in both flow and rain and in MAR-DRIFT plots, the land-use periods analysed in Chapter 3 and the water-resource periods defined by contruction and operationalization of the major storage dams in the basin

<table>
<tr><th>Inflection point periods</th><th>Land-use periods from Chapter 3</th><th>Water resource periods</th></tr>
<tr><td>1959-1971</td><td>1959-1965</td><td rowspan="2">1959-1980</td></tr>
<tr><td>1972-1978</td><td>1966-1975</td></tr>
<tr><td>1979-1989</td><td>1976-1985</td><td rowspan="3">1981-2002</td></tr>
<tr><td>1990-1995</td><td>1986-1995</td></tr>
<tr><td>1996-2002</td><td>1996-2005</td></tr>
<tr><td>2003-2011</td><td rowspan="2">2006-2015</td><td>2002-2007</td></tr>
<tr><td>2012-2016</td><td>2008-2016</td></tr>
</table>

Each of these point periods were analysed to see whether these coincided with significant changes in MAR, DRIFT indicators, rainfall and/or land-use. The mean values for the DRIFT indicators for the periods demarcated by the inflection points are given in Table 4.9. Major changes in MAR between 1979 and 2011, a significant increase between 1979-1989 and 1990-1995 was followed by a progressive reduction in MAR during the following two periods, this was mainly through shorter wet seasons with reduced flows, and longer dry seasons with higher flows (Table 4.10).

Table 4.9 The mean values for DRIFT flow indicators for each period at G1H004-77. Highlighted values show where values are significantly different from the *following* period. Units are given in Table 4.6

	1959-1971	1972-1978	1979-1989	1990-1995	1996-2002	2003-2011	2012-2016
MAR	3.91	5.77	4.48	6.60	4.95	3.61	2.70
Do	30.54	32.85	28.45	32.83	29.43	32.56	37.40
Dd	279	267	281.18	242.16	285.85	266	267
Dq	0.11	0.14	0.36	1.25	0.52	0.38	0.24
Ddv	0.22	0.20	0.31	0.37	0.33	0.28	0.13
Fo	25.69	23.29	22.36	21.83	22.86	24.89	31.40
Fd	17.76	49.57	30.82	72.33	36	43.33	43.40
Fq	38.03	67.22	26.95	72.65	37.34	22.40	24.74
Fdv	1.32	1.93	0.86	1.47	1.28	0.79	0.88
Fv	22.23	73.28	23.81	94.83	37.42	28	34.20
T1dv	0.67	2.75	0.78	0.51	0.55	0.33	0.81
C1w	1.23	1	1.91	1.17	1.86	2	1.20
C2w	2.08	2.71	3.91	2.17	3.14	2.22	1.80
C3w	3.15	2.86	3.45	2.16	4.71	1.88	2.40
C4w	1.54	2	1.64	2.66	1.42	1.22	0.60
C5	1.92	2.42	0.36	1.17	0.714	0	0.20
Rain	808.97	917.17	926.24	992.83	1025.95	631.66	817.13

Table 4.10 Significant differences in aspects of the flow regime between demarcated periods at G1H004-77. Arrow up = significant increase, arrow down = significant decrease while a dash = no significant change

Period	MAR	Do	Dd	Dq	Ddv	Fo	Fd	Fq	Fdv	Fv	C1w	C2w	C3w	C4w	C5	Rain
1959-1971 vs. 1972-1978	-	-	-	-	-	-	↑	↑	-	↑	-	-	-	-	-	-
1972-1978 vs. 1979-1989	-	↓	-	↑	↑	-	-	↓	↓	↓	-	-	-	-	↓	-
1979-1989 vs. 1990-1995	↑	-	↓	↑	-	-	-	↑	↑	↑	-	-	-	-	-	-
1990-1995 vs. 1996-2002	↓	-	↑	↓	-	-	-	↓	-	↓	-	-	↑	↓	-	-
1996-2002 vs. 2003-2011	↓	-	-	-	-	-	-	↓	-	-	-	-	↓	-	↓	↓
2003-2011 vs. 2012-2016	-	-	-	-	↓	-	-	-	-	-	-	-	-	-	-	-

Relative to 1959-1971, mean Fq was double that in the preceding period, Fd was ~32 days longer and Fdv was ~51m^3/s higher in 1972-1978 (Table 4.9). These changes coincided with an increased area of agricultural land. In the next period (1979-1989), marked by the construction of Theewaterskloof Dam and associated works, and the subsequent Theewaterskloof summer irrigation releases into the Berg (around 1980) and winter transfers back to the Theewaterskloof, the Do was five weeks earlier, Dq increased from ~0.14 to ~0.36 m3/s, and Ddv from ~0.19 to ~0.31 Mm^3. The wet season indicators, Fq, Fdv and Fv were all significantly lower in 1979-1989 relative to 1972-1978, and the frequency of the 1:2 year floods reduced dramatically. Relative to 1979-1989, MAR, Fq and Fv were all significantly lower in 1990-1995, despite no significant change in rainfall. Dd was significantly longer (~14 days) and the Dq was significantly higher, reflecting increased summer releases for irrigation. The rainfall in 1996-2002 was higher than in the preceding period but not significantly so. Notwithstanding this, MAR, Fq, Fv and C4w all significantly declined. The dry season (Dd) was significantly longer and Dq was significantly lower. In contrast, C3w floods almost doubled in number in the latter period, suggesting that the higher rainfall influenced this size-class of floods but was not reflected in a general increase in flow. The area of agricultural land-use decreased, and the number of farms increased (from ~40 to more than ~120 dams) over this time (Figure 4.17). 2003-2011 was a particularly dry period, and rainfall was significantly lower (~632 mm) than in 1996-2002 (~1026 mm). This was reflected in MAR, Fq, the number of C3w floods, and the number of C5 floods, all of which were significantly lower than in the earlier period (Table 4.9). The flows in 2003-2011 and 2012-2016 were similar, and only Ddv was significantly lower in the second period, because after the construction of the Berg River Dam and Supplement Scheme, irrigation releases were made from the Supplement further downstream than they had previously been (to avoid unnatural increases in dry season flow in the river reach measured by G1H004-77).

As expected the pattern of changes for the inflection periods closely matched those for the similar land-use periods from Chapter 3 (Table 4.11).

Table 4.11. Average values of DRIFT indicators for land-use (LU) periods at G1H004-77. Highlighted values show where values are significantly different from the *following* period. Units are given in Table 4.6

	LU period 1	Intermediate	LU period 2	Intermediate	LU period 3	LU period 4
	1959-1965	1966-1975	1976-1985	1986-1995	1996-2005	2006-2015
MAR	3.82	4.33	5.08	5.92	4.42	3.32
Do	32.29	30.80	29.30	31	31.10	34.19
Dd	266.29	281.90	277.90	258	273.10	272.64
Dq	0.08	0.13	0.28	0.92	0.46	0.33
Ddv	0.21	0.19	0.29	0.35	0.30	0.22
Fo	24.85	26.30	22.40	21.30	24.50	26.90
Fd	16.42	29.80	31.40	62.40	29.20	51.54
Fq	40.60	44.02	42.05	55.25	32.50	23.70
Fdv	1.44	1.37	1.23	1.28	1.14	0.81
Fv	23.42	39.30	35.50	72.10	29.50	35.45
T1dv	0.53	1.22	1.82	0.53	0.48	0.54
C1w	1	1.40	1.70	1.30	1.80	1.72
C2w	2	2.50	3.10	3.20	3	1.90
C3w	2	4	2.50	3.30	3.70	2.37
C4w	1.71	1.50	1.80	2.30	1.50	0.81
C5	1.85	2.10	1.10	0.80	0.50	0.09
Rain	840.44	797.40	945.26	977.23	914.01	686.44

The water-resource periods (Table 4.8) are defined by two major water storage dams. The Theewaterskloof-Berg Scheme came on-line in 1980 and the first two water resource periods are before (1959-1980) and after (1981-2000) the Theewaterskloof-Berg Scheme started releasing water into the Berg River. The second two periods are before and after the Berg River Dam was completed in 2007/08. Table 4.12 gives the average values for DRIFT indicators at G1H004-77 before and after these two developments.

After implementation of the Theewaterskloof Berg Scheme, Ddv and Dq increased from 0.21 to 0.34 Mm^3, and 0.13 to 0.66 m^3/s, respectively. The number of within-year floods also increased (not significantly), but the 1:2 year floods (C5) occurred significantly less often, from one every 1.95 years to one every 0.68 years. The increase in dry season discharge may be directly attributable to the summer Theewaterskloof transfers to the Berg. The changes in the floods, however, are probably linked to rainfall, which was significantly higher in the after period.

After implementation of the Berg River Dam there was little or no change in the DRIFT (Table 4.12), although as mentioned for the inflection point periods, Dq decreased significantly, reflecting the change in the location of irrigation releases (further downstream). The decreases in floods may reflect the lower rainfall experienced after the completion of the dam; a general reduced variability characteristic of river downstream of large impoundments; and/or changes in land-use.

Table 4.12. Average values for DRIFT indicators at G1H004-77 before and after the Theewaterskloof-Berg Scheme and the Berg River Dam. Highlighted values are significantly different between the two periods. Units given in Table 4.6

Indicators	Theewaterskloof-Berg Scheme		Berg River Dam	
	Before	After	Before	After
	1959-1980	1981-2000	2001-2007	2008-2016
MAR	4.49	5.28	4.12	3.19
Do	31.68	29.36	31.56	35
Dd	271.77	275.64	262.56	273.33
Dq	0.14	0.66	0.53	0.30
Ddv	0.21	0.35	0.29	0.22
Fo	24.86	22.14	23.89	28.22
Fd	29.59	43.27	49.33	42.89
Fq	45.97	43.07	26.52	24.13
Fdv	1.48	1.15	0.80	0.88
Fv	38.59	47.55	36.89	32.67
T1dv	1.32	0.65	0.35	0.57
C1w	1.14	1.77	1.78	1.67
C2w	2.36	3.27	3	1.67
C3w	3	3.59	3.22	2.11
C4w	1.59	1.95	1.44	0.78
C5	1.95	0.68	0.11	0.11
Rain	843.06	987.13	842.20	687.20

4.4.2.2 Lower foothills sub-basin (G1H020 at Daljosafat)

Double-mass plots were constructed for the data from G1H020 for the period 1950 to 2017 (Appendix Figure 4). Different indicators showed change (inflection points) in different years but in most cases inflection points agreed to within a year or two. The inflection points in the MAR-DRIFT indicator relationships that coincided with those in the flow-rain relationships were: 1970/71, 1975/76, 1979/80, 1983/84, 1994/95, 1997/98, 2002/03, 2005/06, 2012/13 and 2015/16 (Figure 4.18). Figure 4.18 shows the three-year average dry season volume; the area under agriculture in the lower foothills; and the number of dams upstream of the gauge to give some idea of changes that occurred.

The mean values for the DRIFT indicators for the periods demarcated by double-mass plot inflection points are given in Table 4.13 and Table 4.14. As was the case for G1H004-77, these are very close the land-use periods analysed in Chapter 3 (Table 4.13).

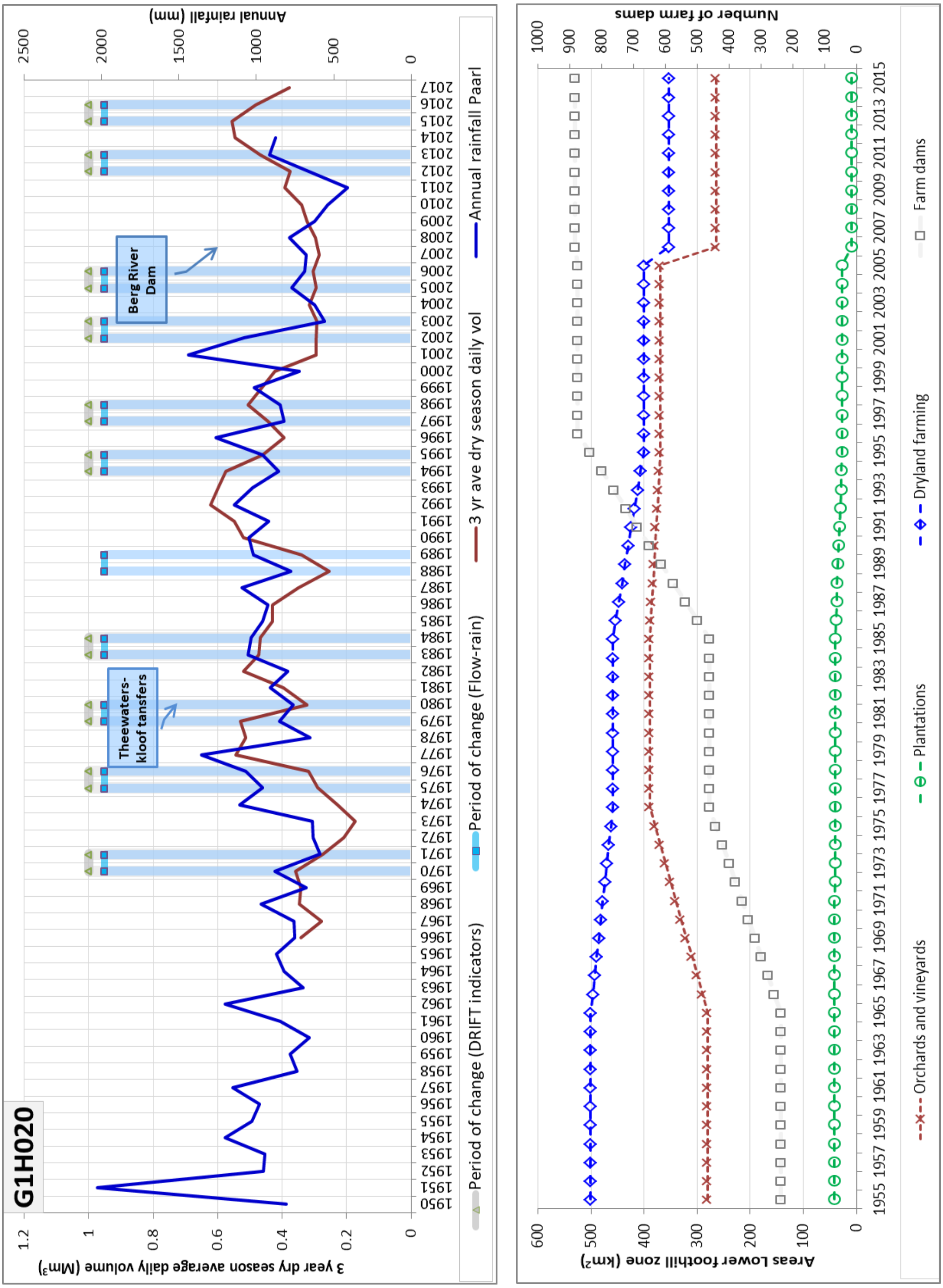

Figure 4.18 **(Left) Periods of change as determined by MAR-DRIFT indicators and the flow-rain relationship, 3-year average annual volume at G1H020, rainfall; (Right) Area of agricultural land-use, and number of dams in the lower foothills**

Table 4.13 G1H020 data: Periods where inflection points occurred in both flow and rain and in MAR-DRIFT plots, the land-use periods analysed in Chapter 3 and the water-resource periods defined by contruction and operationalization of the major storage dams in the basin

Inflection point periods	Land-use periods from Chapter 3	Water-resource periods
1966-1969	1959-1965	1966-1980
1970-1974	1966-1975	
1975-1979	1976-1985	1981-2002
1980-1983		
1984-1988	1986-1995	
1990-1995		
1995-1997	1996-2005	
1998-2002		
2003-2005	2006-2015	2002-2007
2006-2012		2008-2016
2013-2016		

Table 4.14 Average values for DRIFT indicators for each period at G1H020. Highlighted values show where values are significantly different from the *following* period. Units are given in Table 4.6

	1966-1969	1970-1974	1975-1979	1980-1983	1984-1988	1990-1995	1995-1997	1998-2002	2003-2005	2006-2012	2013-2016
MAR	8.33	7.50	12.36	8.98	11.48	13.37	11.03	10.50	7.07	10.09	11.28
Do	34.75	37.60	36.80	32.75	37.80	33.50	36.33	36.60	37	37.43	38.75
Dd	251.50	257.60	240.60	259	218.40	240.33	278.33	234.80	272.33	235.71	204.25
Dq	0.20	0.06	0.36	0.48	0.42	0.98	0.53	1.22	0.73	1.09	1.48
Ddv	0.34	0.24	0.43	0.47	0.35	0.56	0.44	0.37	0.29	0.34	0.51
Fo	26.25	31.60	25.60	26.50	23.60	23.50	26.33	23.80	29.67	25.86	30.50
Fd	51.50	38.80	57	34	84.80	64.17	53	85	39	76.43	33.75
Fq	72.63	49.08	113.94	77.80	94.40	149.52	123.53	90.36	75	78.11	95.50
Fdv	2.26	2.01	3.20	2.36	2.33	3.56	3.33	2.35	2.75	1.90	2.47
Fv	110.75	98.40	199.80	102.25	204.20	216.50	156	200.60	90.67	167.14	150.50
T1dv	1.17	1.40	2.61	2.05	1.70	1.46	3.68	1.04	1.01	1.38	0.71
C1w	3.25	4	2.20	3.25	1.40	2.50	2.33	3.60	2.67	3.86	4
C2w	5.50	3.80	2.80	2.50	4.0	1.8	2.67	3	2	4.42	3.50
C3w	3.50	2.40	3.20	4.50	3.40	2.5	6	3.20	2.67	2.71	1.50
C4w	1.25	1.80	2.20	1.25	3.40	3.33	2	2.20	1.67	0.71	1.75
C5	0	0.40	1.20	0.50	0.80	1	0.33	1	0.67	1.14	1
Rain	827.08	685.02	1029.40	830.20	1014.50	988.59	1027.30	968.49	753.80	643.03	817.13

In general, Dq has been consistently higher since around 1980 relative to earlier times, while other indicators show more varied changes (Table 4.14). The incease of Dq was also reflected in the changes between land-use periods from Chapter 3 (Table 4.15).

Table 4.15 Average values for DRIFT indicators for land-use periods at G1H020. Highlighted values show where values are significantly different from the *following* period. Units are given in Table 4.6

	Intermediate	LU period 2	Intermediate	LU period 3	LU period 4
	1966-1975	1976-1985	1986-1995	1996-2005	2006-2016
MAR	8.29	10.97	12.27	9.76	10.51
Do	36.60	34.90	34.70	37.30	37.90
Dd	250.90	244.60	243.10	250.70	224.27
Dq	0.12	0.43	0.77	0.95	1.22
Ddv	0.28	0.46	0.45	0.36	0.40
Fo	28.20	26.80	23.50	25.80	27.54
Fd	51.60	45.10	69	67.40	60.90
Fq	64.82	97.99	123.12	97.40	84.43
Fdv	2.15	2.77	3.13	2.69	2.10
Fv	122.10	147.30	205.10	166.10	161.09
T1dv	1.59	2.02	1.57	1.75	1.13
C1w	3.30	2.60	2.40	2.90	3.90
C2w	4.40	2.70	2.80	2.60	4.10
C3w	3	3.10	3.60	3.70	2.30
C4w	1.70	2.10	3.10	2	1.10
C5	0.40	0.90	0.70	0.80	1.10
Rain	797.41	945.26	977.27	914.01	686.44

At this gauge, there were 15 years before Theewaterskloof (1966-1980) and, as for G1H004-77 (1981-2000; Table 4.13). After the commencement of transfers, there were significant increases in rainfall, Dq, and C4w floods, and the wet season (Fo) started three weeks earlier. In contrast, there were significantly fewer C2w floods (Table 4.16).

The years for the Berg River Dam are for the same years as those in G1H004-77. MAR and Ddv increased significantly after closure of the dam, despite a slightly (but not significantly) lower rainfall, and there were significantly fewer C3w floods (Table 4.16).

Table 4.16 Averages for DRIFT indicators before and after the Theewaterskloof-Berg Scheme and the Berg River Dam at G1H020. Significant differences are highlighted. Units given in Table 4.6

	Theewaterskloof-Berg Scheme		Berg River Dam	
	Before	After	Before	After
	1966-1980	1981-2000	2001-2007	2008-2016
MAR	9.30	11.46	9.54	11.72
Do	36.33	35.41	37.67	38.22
Dd	251.40	241.14	238.44	223.44
Dq	0.22	0.78	1.14	1.19
Ddv	0.34	0.44	0.30	0.43
Fo	27.80	24.41	25.56	28.67
Fd	46.07	69.23	76.67	52.67
Fq	78.27	110.65	85.20	84.81
Fdv	2.52	2.80	2.46	2.15
Fv	130	190.18	174.11	155.11
T1dv	1.72	1.84	1.16	1.02
C1w	2.93	2.73	3.11	3.78
C2w	3.87	2.77	3.44	3.67
C3w	2.93	3.73	3.33	1.89
C4w	1.73	2.64	1.89	1.22
C5	0.53	0.82	0.89	1.11
Rain	844.28	987.13	842.20	687.20

4.4.2.3 Lower foothills sub-basin (G1H036 at Vleesbank)

Double-mass plots were constructed for the G1H036 data for the period 1950 to 2017 (Appendix Figure 4). The DRIFT-MAR inflection points that coincided with those in the flow-rain graphs are plotted in Figure 4.19, *viz*.: 1982/83, 1987/88, 1992/93 and 2010/11. Figure 4.19 also shows the three-year average dry season volume, the agricultural area in the lower foothills, and the number of dams upstream to give an idea of underlying changes. Once again, these are very close the land-use periods analysed in Chapter 3 (Table 4.17). As was the case for the upstream gauges, the most noticebale of the outcomes was a post-1980 an increase in Dq linked to irrigation releases and an increase in Dd, linked to abstraction in general.

Thre closely matched the changes for land-use periods, although in these the earler onset of Fo and the reduction in Fq was more clear (Table 4.19). In common with the other gauges, these changes are clearly not linked to differences in rainfall between the various periods.

The effects of the Theewaterskloof-Berg River Scheme could not be assessed at G1H036 as the record started in 1979, only one year before the start of the transfers. Comparison for data for before (2002-2007) and after (2008-2013) the Berg River Dam showed that Dq increased significantly (more than doubled) although rainfall did not change significantly.

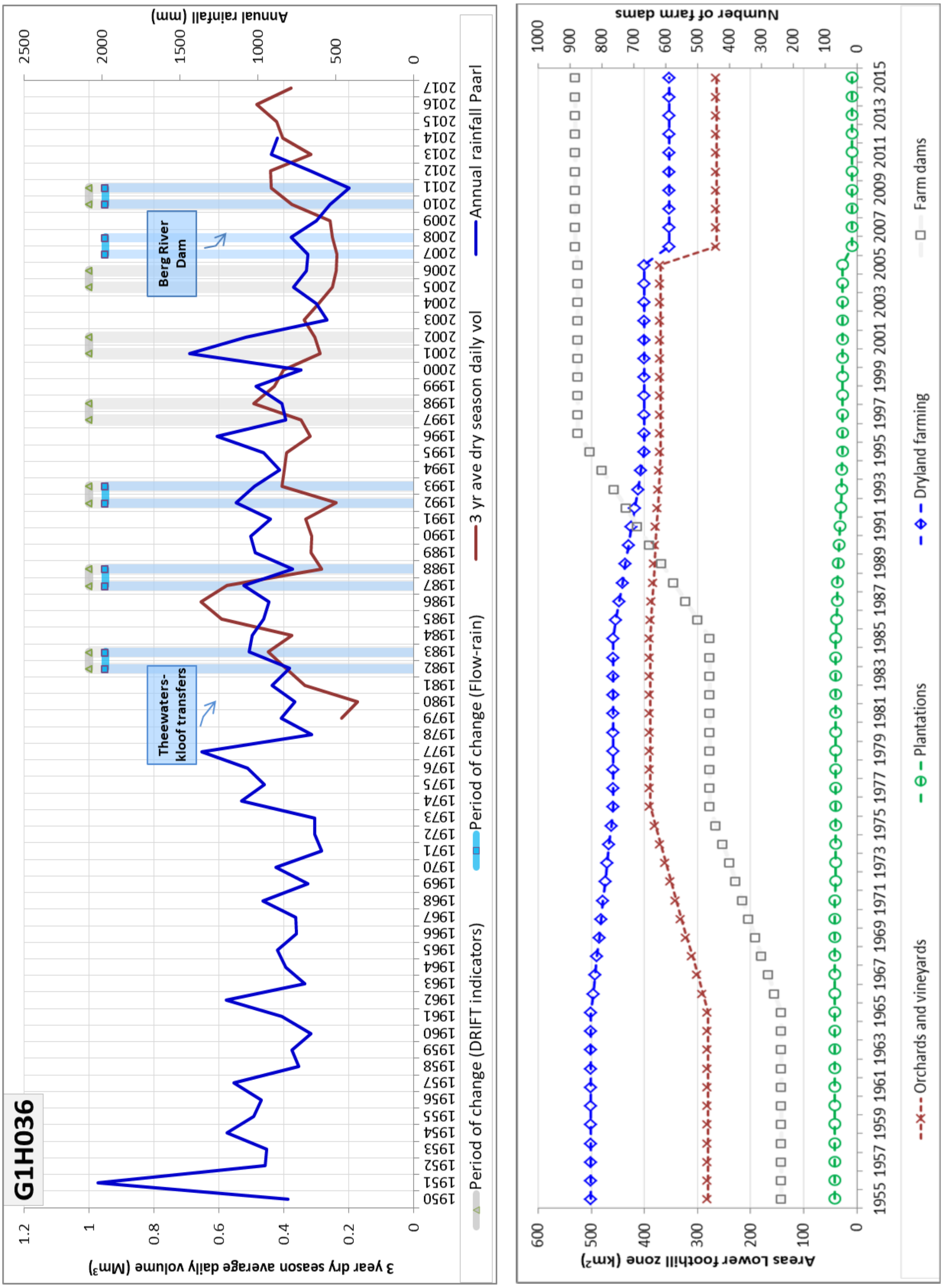

Figure 4.19 **(Left) Periods of change as determined by MAR-DRIFT indicators and the flow-rain relationship, 3-year average annual volume at G1H036, rainfall; (Right) Area of agricultural land-use, and number of dams in the lower foothills**

Table 4.17 G1H036 data: Periods where inflection points occurred in both flow and rain and in MAR-DRIFT plots, the land-use periods analysed in Chapter 3 and the water-resource periods defined by contruction and operationalization of the major storage dams in the basin

Inflection point periods	Land-use periods from Chapter 3	Water-resource periods
-	1959-1965	1966-1980
-	1966-1975	
1979-1982	1976-1985	1981-2002
1983-1987		
1988-1992	1986-1995	
1993-2006	1996-2005	
2007-2013	2006-2015	2002-2007

Table 4.18 Average values for the DRIFT indicators for each period for G1H036. Highlighted values show where values are significantly different from the *following* period. Units are given in Table 4.6

	1979-1982	1983-1987	1988-1992	1993-2006	2007-2013
MAR	9.50	14.02	13.22	9.28	9.26
Do	39.75	34.20	38.80	36.50	37.43
Dd	149	222.40	214.20	255.79	241.57
Dq	0.50	0.10	0.10	0.27	0.54c
Ddv	0.35	0.49	0.29	0.35	0.33
Fo	25.50	21.20	22.60	25.36	24.29
Fd	74.50	99.20	81.60	71.71	73
Fq	86	140.40	122.04	80.14	79.66
Fdv	2.66	2.93	3.01	2.34	1.92
Fv	159.50	279.40	239	168.21	179
T1dv	0.84	2.82	1.61	1.31	1.01
C1w	3.50	1.60	2.20	3.07	2.43
C2w	2.50	4.20	5.20	6.50	5.86
C3w	2.25	4	3.20	2.29	1.86
C4w	1.25	2.20	2	1.36	0.86
C5	0.50	1.60	1.40	0.86	1.14
Rain	830.23	1014.58	981.42	905.32	659.17

Table 4.19 Average values of DRIFT indicators for land-use periods at G1H036. Highlighted values are significantly different from the *following* period. (Units in Table 4.6), years in brackets indicate the LU period according to Chapter 3, where the gauged record does not include the full period

	LU period 2	Intermediate	LU period 3	LU period 4
	(1976) 1979 -1985	1986-1995	1996-2005	2006-2013 (2015)
MAR	11.64	12.44	9.06	9.06
Do	37.14	36.80	36.90	37.50
Dd	190.14	219.70	258.60	242.13
Dq	0.31	0.18	0.26	0.50
Ddv	0.45	0.33	0.34	0.32
Fo	24	21.80	26.50	24.38
Fd	71.43	94.40	68.40	73.63
Fq	121.94	117.47	68.76	77.48
Fdv	2.89	2.86	2.25	1.89
Fv	184.86	256.60	159	173.50
T1dv	2.38	1.41	1.18	1.05
C1w	2.86	2.50	2.70	2.63
C2w	2.28	5.80	6.50	6.25
C3w	2.57	3.20	2.50	1.88
C4w	1.43	2.10	1.30	0.88
C5	1	1.30	0.90	1
Rain	910.43	977.27	914.01	663

Table 4.20 Average values of DRIFT indicators before and after the Berg River Dam at G1H036. Highlighted values are significantly different from each other. Units given in Table 4.6

	Before	After
	1999-2007	2008-2013
MAR	9.01	8.83
Do	37.44	37.50
Dd	252.56	242.50
Dq	0.24	0.60
Ddv	0.27	0.35
Fo	26.33	24.33
Fd	76.22	67.83
Fq	75.13	73.83
Fdv	2.18	1.81
Fv	178	164.50
T1dv	1.10	0.94
C1w	3.33	2.33
C2w	6.56	5.83
C3w	2.11	1.50
C4w	1.22	0.83
C5	1	1
Rain	842.20	656.07

4.4.2.4 Lowlands sub-basin (G1H013 at Drie Heuwels)

Double-mass plots were constructed for gauge G1H013 data for the period 1963 to 2017 (Appendix Figure 6). Periods where inflection points occurred in both flow and rain and in MAR-DRIFT plot are shown in Table 4.21 and Table 4.22.

Table 4.21 G1H013 data: Periods where inflection points occurred in both flow and rain and in MAR-DRIFT plots, the land-use periods analysed in Chapter 3 and the water-resource periods defined by contruction and operationalization of the major storage dams in the basin

<table>
<tr><th>Inflection point periods</th><th>Land-use periods from Chapter 3</th><th>Water-resource periods</th></tr>
<tr><td>1964-1968</td><td>1959-1965</td><td>1964-1968</td></tr>
<tr><td>1968-1973</td><td rowspan="2">1966-1975</td><td>1969-1973</td></tr>
<tr><td>1974-1978</td><td>-</td></tr>
<tr><td rowspan="2">1978-1994</td><td>1976-1985</td><td rowspan="3">1981-1999</td></tr>
<tr><td>1986-1995</td></tr>
<tr><td>1995-1999</td><td>1996-2005</td></tr>
<tr><td>2000-2009</td><td rowspan="2">2006-2015</td><td>1999-2007</td></tr>
<tr><td>2009-2013</td><td>2008-2015</td></tr>
</table>

Table 4.22 Average values for DRIFT indicators and rain for each period for G1H013. Highlighted values are significantly different from the *following* period. Units are given in Table 4.6

	1964-1968	1969-1973	1974-1978	1978-1994	1995-1999	2000-2009	2010-2015
MAR	13.74	12.22	20.75	19.20	16.98	18.37	15.82
Do	36	36	39.50	35.17	32	33.60	34.83
Dd	197.20	235.80	152.25	228.06	197.60	271	250.83
Dq	0.70	0.48	2.08	0.48	2.48	1.07	1.22
Ddv	0.56	0.43	0.66	0.66	0.71	1.12	0.57
Fo	23.60	26.40	16	21.59	22.60	23.80	24.17
Fd	68.60	35.60	123.25	69.65	50.80	44.10	67
Fq	97.40	101.58	172.18	216.99	161.12	200.29	144.12
Fdv	4	3.88	4.07	5.74	5.95	5.39	4.21
Fv	225	167.80	449.50	355.82	192.40	223	330.83
T1dv	1.56	2.48	2.12	2.48	2.26	1.87	2.56
C1w	0.80	1.20	1	0.71	1	1.20	1.17
C2w	4.40	3	1	2.24	2	2.80	2.50
C3w	4	3.60	2	2.65	2.80	3.20	3
C4w	0.40	0.20	1.50	2.41	3	2.20	2.50
C5	0.40	0.80	2.50	1.29	1	1.30	0.83
Rain	834.82	685.02	1123.35	931.64	980.66	797.40	680.48

Many of the changes in the record at G1H013 appear to be linked to rainfall and/or an increase in water to the basin. For instance, an increase in C3w, and Fq, Fv, Fd, C4w, Dq, and Ddv all unceased between periods, although not always significantly so (Table 4.22). The same is true for the periods characterised by land-use change (Table 4.23).

Table 4.23 Average values of DRIFT indicators for Chapter 3 land-use (LU) periods at G1H013. Highlighted values are significantly different from the *following* period. Units given in Table 4.6

	Intermediate	LU period 2	Intermediate	LU period 3	LU period 4
	1966-1975	1976-1985	1986-1995	1996-2005	2006-2015
MAR	13.68	17.73	22.81	14.27	19.05
Do	36.50	33.70	37.90	34.20	32.50
Dd	219.08	198.50	213.50	239.30	271.20
Dq	0.59	1.01	0.73	1.48	1.35
Ddv	0.47	0.83	0.57	0.74	1
Fo	24.33	20.80	20.50	24.20	23
Fd	53.33	72.10	85	51.60	56.80
Fq	111.75	173.11	255.67	137.44	204.25
Fdv	4.19	5.10	6.29	4.70	5.11
Fv	205.58	313.60	434.90	201.90	301.70
T1dv	2.20	1.21	3.26	1.87	2.63
C1w	1	1	0.50	1.40	0.90
C2w	3.42	2.20	1.80	2.50	2.60
C3w	3.58	2.30	2.60	3	3.30
C4w	0.33	1.90	3.30	2.20	2.40
C5	0.92	1.30	1.70	0.70	1.30
Rain	805.57	945.26	977.27	914.01	686.44

The influence of the water-resource developments on the flows at G1H013 are less clear than at the upstream gauges. Relative to the period immediately before its construction, after Voelvlei Dam was constructed, there was a significant decrease in intermediate size intra-annual flows (C2w and C3w), and an increase in C5 (1:2 year) floods (Table 4.24). Relative to before, the period immediately after Theewaterskloof came on line, Fq and Fdv were significantly higher and there were more frequent C4w floods, for the Berg River Dam; Dq increased significantly from 0.98 to 1.35 m^3/s.

Table 4.24 Statistical differences in DRIFT indicators before and after the Theewaterskloof-Berg Scheme, Berg River and Voelvlei Dams at G1H013. Highlights indicate significant differences. Units given in Table 4.6

	Voelvlei Dam		Theewaterskloof-Berg Scheme		Berg River Dam	
	Before	After	Before	After	Before	After
	1964-1970	1971-1980	1971-1980	1981-1999	1999-2007	2008-2015
MAR	13.37	17.03	15.14	20.52	17.32	16.59
Do	37.29	36.29	35.67	34.67	35.11	33.88
Dd	209.57	186.71	204.33	220.06	263.22	259.25
Dq	0.61	1.41	0.86	1.03	0.98	1.35
Ddv	0.49	0.59	0.58	0.70	0.85	0.91
Fo	23.86	21.00	21.67	21.22	24.56	23
Fd	55.86	89.00	71.67	70.44	51	61.63
Fq	89.49	151.03	135.88	230.03	195.69	147.44
Fdv	4.03	3.93	4.02	6.23	5.31	4.17
Fv	185.71	351.71	283.78	362.28	230.78	294.38
T1dv	1.97	2.13	1.84	2.61	1.62	2.98
C1w	0.57	1.43	1.22	0.56	1.44	1.13
C2w	4.14	1.71	3.05	1.94	2.56	2.75
C3w	4.42	2.14	1.89	2.89	3.11	3.13
C4w	0.43	0.86	0.89	2.67	2.44	2.25
C5	0.28	2.00	1.00	1.44	1.11	1
Rain	819.61	907.70	873.22	970.16	842.20	687.20

4.4.3 Summary of results

The records show clearly that the volume and distribution of flows in the Berg River have changed progressively over time in a manner that is relatively independent of rainfall. In general, flow in the upper foothills (G1H004-77) showed more frequent and significant changes than did the other gauges, but the some changes are apparent along the full length of the river. There was a clear and general trend of increasing Dq (dry season daily discharge) and/or Ddv (dry season average daily volume), decreasing Fq (flood season daily discharge) and/or Fdv (floods season average daily volume), accompanied by decreasing C1w to C5 floods (the four within year flood classes and the 1:2 year floods) (Figure 4.8 to Figure 4.14). The causes of some of these changes, such as increased Dq, are clearly linked to the major water-resource developments, but the reasons for other changes are less clear.

The increases in Dq and/or Ddv are clearly associated with irrigation releases from the major dams. When the Theewaterskloof-Berg River transfer scheme started operating, Dq increased significantly all the way downstream, with more marked differences shown in the upper and lower foothills (Figure 4.20). After the completion of the Berg River Dam, a significant decrease in Dq was seen at G1H004-77 (returning to Dqs closer to the pre-Theewaterskloof situation), however the gauges further downstream continued to show increases relative to pre-Theewaterskloof, and relative to pre-Berg Dam. The return to conditions closer to pre-Theewaterskloof at G1H004-77 is because the irrigation releases were tunneled to a point further downstream, below the gauge. The increases in Dq/Ddv further downstream are due to transfers and releases from these two developments as well as other irrigation releases (for example, releases from Voelvlei).

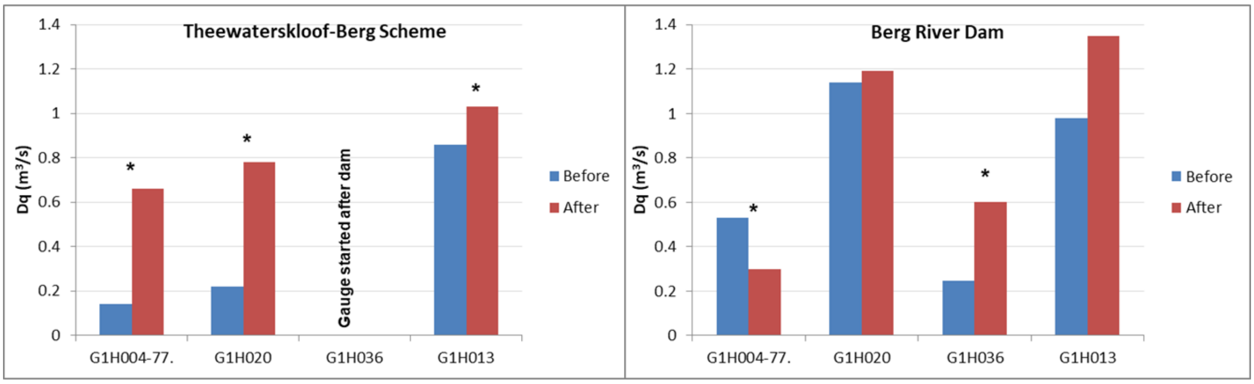

Figure 4.20 Differences in Dq before and after the Theewaterskloof-Berg River scheme and the Berg River Dam completion. Asterisk (*) indicate difference is significant

The number of small dams increased in all sub-basins, with the greatest increase in the mid-90s (1995/96). Assuming linear increases in the "interim" land-use periods, there are significant correlations between the increasing farm dams in the upper foothills and increasing Dq and Ddv at G1H036 (Table 4.25); a reduction in frequency of C3w, C4w (Table 4.25); as well as a reduction in the total number of C1w to C5 floods (not shown).

Table 4.25 Correlations between the number of (farm) dams (from Chapter 3) and the main flow indicators

	G1H036		G1H020		G1H036		G1H020	
No. of dams versus	Significance	Direction	Significance	Direction	Significance	Direction	Significance	Direction
Do	-	Positive	-	Positive	-	Positive	-	Negative
Dd	-	Negative	-	Negative	<0.05	Positive	<0.05	Positive
Dq	<0.05	Positive	<0.05	Positive	-	Positive	<0.05	Positive
Ddv	<0.1	Positive	-	Positive	-	Negative	-	Positive
Fo	-	Positive	-	Negative	-	Positive	-	Positive
Fd	-	Positive	-	Positive	-	Negative	-	Negative
Fq	-	Negative	-	Positive	<0.1	Negative	-	Positive
Fdv	-	Negative	-	Positive	-	Negative	-	Positive
Fv	-	Positive	-	Positive	-	Negative	-	Negative
T1dv	-	Negative	-	Negative	-	Negative	-	Positive
T2dv	-	Negative	-	Positive	-	Negative	-	Negative
C1 floods	-	Positive	-	Negative	-	Negative	-	Positive
C2 floods	-	Negative	-	Negative	-	Positive	<0.1	Negative
C3 floods	<0.1	Negative	<0.1	Negative	<0.05	Negative	-	Negative
C4 floods	<0.05	Negative	-	Negative	<0.1	Negative	<0.05	Positive
C1 to C5	<0.05	Negative	<0.05	Negative	-	Negative	-	Positive

There is a similar pattern in the lower foothills at G1H020, although the correlations with Ddv and C4w are not statistically significant. Further downstream in the lower foothills at G1H036, the number of farm dams correlated significantly with a longer Dd, and reduced numbers of C3w and C4w floods. In addition there is a significant correlation with reduction in Fq.

Finally, in the lowlands at G1H013, Dd and Dq have significant positive relationships with the number of farm dams. The correlations with floods is mixed, with C2w having a negative correlation and C4w having a positive correlation. These effects and relationships are similar to those seen for the larger dams and water resource developments discussed above, and the affects may be conflated. In addition, flood events showed a trend towards shorter durations and slightly higher peaks at G1H013 (Figure 4.15).

While commercial afforestation has been associated with changes in runoff and/or flow patterns (e.g. Scott *et al.* 1998), the extensive pine plantations in the upper foothills, and their subsequent removal from around 1985 onwards could not be clearly associated with specific changes in DRIFT flow indicators, as the changes due to the major water resource developments and/or farm dams overrode those which might have resulted from afforestation or deforestation.

4.5 Discussion

Over time, the hydrology of the Berg River has changed significantly from its natural pattern. Reliable records are only available for the last 50 years, so it is impossible to say exactly how much change has occurred, given that land-use changes, abstractions and manipulations of the river channel started as early as the mid-17th century (Meadows 2003).

There were limitations with respect to the analyses in this chapter, including inadequate details of land-use in specific years because land-use was classified into periods determined by the availability of maps. This was further complicated by the differing record lengths of the flow gauges, which made it difficult to distinguish between the effects of land-use change and those of water resource developments such as farm and large dams. Nonetheless, it is clear that earlier flow data are better correlated with rainfall than those later in the record (e.g., Figure 4.5). The “delinking” of flow from rainfall is attributable to anthropogenic changes, such as land use and water resource developments. It is also known that in a Mediterranean climate such as that of the Berg River, farmers historically and today, store winter water (particularly on tributaries) to have it available for crops in the long dry summer months (Kinawy 1976).

The changes in the Berg River Basin that are most obviously linked to flow changes, are those due to the major water resource developments. The hydrological records show that implementation of the Theewaterskloof-Berg scheme, the Berg River Dam and Voelvlei Dam resulted in clear, statistically-significant changes in the pattern and volume of flows in the Berg River, particularly in the upper regions. Most clearly, there were increases in dry season discharge (Dq and/or Ddv), and lesser decreases in wet season discharge (Fq, Fdv and/or Fv) apart from the lower-most site G1H013. The changes are in line with the release of irrigation flows in the summer dry season, and the attenuation and storage of wet season flows. Changes are apparent at all four sites down the length of the Berg River, although the aspects that change differ from site to site. The differences in affected aspects is partly as a result of the effects of distance, attenuation and the contributions from incremental catchments, but also because of the changes of flow due to supply of irrigation water. For instance, irrigation water is released into the Berg River along with water for the Withoogte Scheme, which is then abstracted from Misverstand Dam weir (DWAF, 1994). Also the difference in Dq of the upper foothills with that of the rest of the basin after the construction of the Berg River Dam. At G1H004-77 the overall dry season minimum discharge was significantly lower after the dam but remained high at other downstream gauges. During the winter period irrigation releases are pumped back into the Theewaterskloof-Berg River system through the Skruifraam/Berg River Supplement Scheme (Water Wheel 2006) that is located downstream the Dwars River confluence. According to DWAF (1994b), tributaries between Berg Dam and Paarl contribute about 150 Mm^3/a, some of which is now captured for storage in the Berg River Dam.

The cumulative effects of smaller dams such as those on farms have not been widely investigated (Hart and Hart 2006; Rivers-Moore *et al.* 2007). In South Africa, there is a strong correlation between the number of small dams within a basin and severely reduced low flows, deterioration in various physico-chemical properties (particularly total dissolved salts) of the river, and knock-on effects of biota such as macroinvertebrates (Mantel *et al.* 2010). Other studies have reported that small dams resulted in notable changes in habitat structure as a result of sediment trapping (Stanley *et al.* 2002); in species composition as a result of disruption of dispersal of species with poor floating capacity (Jansson *et al.* 2000); and reduced density of cold-water fishes, such as trout, and; shifts in macroinvertebrate community composition due to increases in mean summer temperatures below dams (Lessard and Hayes 2003). Rosenberg *et al.* (2000) also demonstrated that a large number of and area covered by small dams can have significant effects. In this study, it was, however, not possible to distinguish clearly between flow changes linked to farm dams and those linked with large water resource developments and land-use changes, which appeared to have a far greater impact in this catchment. Likewise, the effects of afforestation and deforestation in the upper foothills of the Berg River Basin could not be separated or determined due to the overriding influences of the major water resource developments and/or farm dams.

The results from Chapter 3 indicate that the area under cultivation in the basin began declining from LU period 2 (1975-1985), and has continued this downward trend to the present (2015). In this study, areas previously under cultivation are now classified as “fallow” and/or “natural vegetation”. However, this decrease in cultivated land is not necessarily associated with a decrease in water consumption on those farms because there has been a switch to bed and breakfasts and guesthouses, which may also use a lot of water. A recent survey of agricultural land purchased between 2005 and 2007 in the Cape Winelands area of the Western Cape reported that more than half were lifestyle buyers (Reed and Kleynhans 2009) who were less dependent on farming income, and thus less concerned about realising the agricultural productivity of the land. However, water availability, including availability of irrigation infrastructure, river access and the presence of farm dams were among the most important characteristics of consideration to the buyers. Thus, while the increase in purchase of agricultural lands for alternative uses may have contributed to the decrease of land under cultivation/irrigation, this does not imply a reduction in water consumption. Indeed, it may be the reverse, with more water being used to maintain and run farms where luxury and aesthetics are important feature, and were tourist accommodation is created (with concomitant high water use).

5 Historical changes in channel planform of the Berg River

5.1 Introduction

The aim of this chapter is to give an overview of the historic Berg River planform using historic images and maps in order to document the degree of change over time.

A river's channel shape and its flow, sediment, chemical and thermal regimes are highly structured (e.g. Leopold and Wolman 1957; Bradshaw 1978; Rosgen 1994). Left on their own, river channels change naturally from source to sea in fairly predictable ways (Bradshaw 1978). They are steeper and often situated in steep-sided valleys in mountainous areas, and in flatter terrain they are typically low-gradient channels, with or without floodplains, in wider valleys (Bethune 2009). Along the way, the character of a river transforms through a series of zones largely driven by gradient and distance from origin: source, mountain torrent, mountain stream, foothill, transitional, lowland, and estuary. Within each of these zones, the morphology of a river channel is a function of a number of processes and environmental conditions, including the composition and erodibility of the bed and banks (e.g. sand, clay, bedrock) and the flow regime (Rowntree *et al.* 2000). Logically, rivers with a large mean annual discharge are expected to have greater cross-sectional areas than those with smaller average flows, but in reality, the width of a river's channel and its riparian zone varies greatly according to local geology, hydrology and aspects such as stream power (Leopold and Wolman 1957; Naiman *et al.* 2005), which is defined as a function of stream power, specific weight of the water, discharge and slope. Thus, rivers tends to have characteristic forms extending over long reaches which, when observed in planview, i.e. from above, display distinct geometric patterns (e.g. Rosgen 1994). These patterns of a river channel are used to describe its form and are generally a graduation from straight, through meandering to braided, with or without riffle/pool sequences (Leopold and Wolman 1957). Rivers run on different slope ranges, starting from very steep (gradient >0.10) to entranched gullies (gradient 0.02 to 0.039). Channel slope is commonly associated to bed features and reach types such as pools, riffles, rapids, cascades, and steps that often alternate through the segment (Grant *et al.* 1990; Rosgen 1994).

Only a limited section or reach of a river is referred to when describing channel patterns (Leopold and Wolman 1957) and the notion of sinuosity is a key feature of these descriptions (Schumm 1963). A straight channel has minimal sinuosity (< 1.1; Dey 2014) at bankfull conditions. Meandering channel have (sinuosity >1.5). Braided channels result from the division of a single trunk channel into a network of small channel branches consisting of small islands (Leoplod and Wolman 1957; Dey 2014). The difference between a sinuous and meandering course is the degree and how symmetrical the successive bends are.

The riparian zone abuts the river channel and is defined as a distinct band of vegetation whose boundary is demarcated on the basis of soil conditions, vegetation and bank topography (Naiman *et al.* 2000). A river's flow regime is the primary determinant of the structure of the vegetation in the riparian zone (Poff and Ward 1989, Gurnell *et al.* 2011) and influences the distribution, diversity and abundance of species (Richter *et al.* 2003; Naiman *et al.* 2005). Most plants in the riparian zone are adapted to cope with conditions associated with flood events such as sediment deposition, physical abrasion, and stem breakage (Busch and Smith 1995; Naiman *et al.* 2005). Different riparian species have different tolerances to floods and droughts, and growth responses to inundation (Friedman and Auble 2000; Pettit *et al.* 2001).

The width of the riparian zone varies greatly according to stream size, local geomorphology, and hydrological regime (Naiman *et al.* 2005). In most headwater streams, the riparian zone is narrow and, because the rivers are also narrow, in some cases can be a closed canopy. In mid-sized rivers, the riparian area is wider, determined by >50 years channel dynamics and annual flow variability. Large rivers often, but not always, have complex floodplains, lateral channel migration and a diverse vegetative community (Naiman and Decamps 1997), particularly in their lower reaches.

The natural variability of the dominant factors that control channel morphology and the nature of the riparian zone, like discharge and sediment load (Leopold and Wolman 1960; Beck and Basson 2003), means that a river channel is never completely stable (Lane and Richards 1997), but rather continually strives to reach an equilibrium state through subtle changes to cross-section, slope and/or channel pattern to uphold optimal transport of water and sediments. River reaches are considered in equilibrium or more-accurately, quasi-equilibrium when there is a balance between the amount of sediment supplied to the system and the capacity of the system to transport that sediment (Field 2002). In the quasi-equilibrium condition, river variables, such as discharge, sediment supply, channel width and depth, are mutually interdependent, meaning that a change in any one parameter will illicit a response in one or more of the others. In reality, however, once a river reaches this quasi-equlibrium state, major changes in channel planform tend to occur only in response to significant events, such as a large flood, or a major disruption in the flow and sediment regime, such as when a dam is constructed upstream or as a result of direct interventions, such as bulldozing of its banks or bed, and/or removal of the riparian vegetation (Beck and Basson 2003). Planform change can also be the result of a more straightened course imposed on the river through different land-use and channel management activities, or a channel response to other adjustment processes such as aggradation and/or widening.

A wide range of direct and indirect human activities affect channel morphology and its riparian area including channelization, dam construction, abstractions, urbanization and other land-use management (Gregory 2006). The complexity of riparian zones and their reactions to driving variables, however, means that it is often difficult to separate the impacts humans from natural influences (Tooth 2000; Rebelo *et al*. 2013).

Typically, highly sinuous and/or braided river channels (and the variety of habitats they offer) pre-date urbanisation or extensive land-use change (Kiss *et al.* 2008). The Tisza River in Hungary was naturally highly sinuous with sharp bends and wide channel, but surveys after extensive basin development showed a straightening of the river and a decrease in channel width (Gurnell *et al.* 1994; Kiss *et al.* 2008). Similarly, the Taro River in the Northern Italian Apennines, decreased channel length, width and braiding resulted from parts of the riparian area and river bed being used for agricultural and industrial purposes (Clerici *et al.* 2015). Urban development also increases channel width and carrying capacity of river channels (Booth 1990). At Watts Branch in Washington D.C, urbanization decrease channel width at first, but after some time, the number of floods that exceeded channel capacity increased (from two per year to more than ten per year) and with this the channel capacity then began to increase (Leopold 1972). Increase in channel width and capacity following urban development has also been reported by Gregory (2006) and Lauer *et al.* (2017). Water resource developments such as such as dams, diversions or abstractions also affect channel morphology resulting in channel incision, channel narrowing and changes in channel pattern (Surian and Rinaldi 2003) with the effects felt closest to infrastructure (Gregory 2006) primarily because these change the relationship between the flow of water and sediment.

Changes resulting from urbansiation, land-use change and/or water-resource developments also affect the riparian vegetation, primarily through knock-on effects related to channel shape change, flood inundation levels, and sediment and nutrient supply (King and Brown 2000). This is because, low and high flows are important for the maintenance of a riparian community (King *et al.* 2003b), and the intensity and duration of flow plays an important role in their survival and growth. Along the San Miguel River in Colorado, USA, plant communities are arranged according to a hydrologic gradient defined by flood frequency (Friedman *et al.* 2006). For instance, there is a *Salix exigua* community in areas flood more frequently than every 2.2 years while *Alnus incana* and *Betula occidentalis* dominated areas that are only flooded every 2.2 to 4.6 years (Friedman *et al.* 2006). Chages in flooding frequency, thus results in changes in the vegetation community. In the Colorado River Delta in Mexico reduced surface flow has led to a reduction in native trees (*Populus fremontii* and *Salix gooddingii*) and an increase in shrubs, mostly *Tamarix* spp., an exotic halophytic shrub. Prolonged drought or flow reductions where driven by climate or by water astractions lead to a lowering of riparian water tables and ultimately mortality in riparian trees (Richardson *et al.* 2007). The change is not always towards reduced vegetation. Repeat photography in the semi-arid winter rainfall region of South Africa showed an increase in extent of riparian vegetation along river channels affected by dam construction, mainly because of the reduced effect of flooding, which tends to clear riparian species from the river channel (Hoffman and Rohde 2011).

River channels and their riparian vegetation occupy a relatively small area in the landscape, but provide irreplaceable ecosystem functions and services (Gregory *et al.* 1991; Naiman and Decamps 1997; Merritt and Wohl 2002). Healthy riparian zones help to maintain channel form by binding soils and strengthening rivers banks (Thorne 1990). The presence of trees and shrubs reduces flow velocity leading to deposition of fine sediments and seeds in these areas (Chaimson 1989; King *et al.* 2003b), which helps to create sandbanks and bars. Riparian vegetation protects river banks and buffers the river against sediments, fertilizers, pesticides and other matter draining into the river from the surrounding landscape (Dosskey *et al.* 2010). Removal of, or a change in, riparian vegetation exposes the river channel and riparian zone to erosion with inevitable consequences for stream morphology and habitat availability (e.g., Davies-Colley 1997).

Quantifying channel changes can be done by measuring a set of channel characteristics, such as width, depth and planform, before and after changes took place (Gregory 2006). Historical images and maps are important in the study of fluvial change since they contain precise information on the position and characteristics of river courses at particular moments in time (Uribelarrea *et al.* 2003), provided care is taken to avoid challenges linked to the use of disparate sets of information (Hooke and Redmond 1989; Uribelarrea *et al.* 2003). The most common challenges are: (i) information is available from an overhead perspective; (ii) images and or maps lack coordinates or detailed planimetric information; (iii) scales and coverage often vary between one section of a river and another, and/or; between years (Uribelarrea *et al*. 2003). Despite the many associated problems, historic images and maps have been used with considerable success since the 1700s to reconstruct changes in a river planform and continue to be used across the world today (Decamps *et al.* 1989; Gurnell *et al.* 1994; Leys and Werritty 1999; Winterbottom 2000; Werrity and Leys 2001; Jamu *et al.* 2003; Galster *et al.* 2008; Clerici *et al.* 2015; Lauer *et al.* 2017). The methods of data capture used in this chapter are in line with the principles of Temane *et al.* (2014), who showed that expert knowledge and rapid characterization of catchments are viable options for assessing siltation risks and for analysing controlling factors at a larger scale with minimum costs and acceptable accuracy. Historical imagery is of importance as quantitative data on landform change over time can be extracted (e.g. Lane *et al.* 2010). Historical images and maps were also used to investigate change in planform of the River Dee (on Welsh-English border), which has been subject to increasing flow regulation over 115 years, between 1876 and 1992. During a period of increased flow regulation (around 1949), a decrease in channel width and mobility was shown downstream (Gurnell *et al.* 1994). Black and white aerial images were used by Jamu *et al*. (2003) to assess the impacts of vegetation cover degradation on fish, soil erosion and flow in the Likanagla River Basin.

This chapter used available images and maps to provide an assessment of historic change in river planform and the nature and extent of the riparian zone along the Berg River, and their possible links to land-use and water-resource developments. The hypotheses tested were:

(1) In general, the changes that occur in river planform as a result of development will tend towards narrower systems with less habitat diversity.
(2) Different land-uses affect the riparian area and river channel structure in different ways.

5.2 Methods

The original intention with this chapter was to resurvey historical cross-sections recorded for the Berg River so that changes in channel shape could be quantified, but this was not possible because the old benchmarks could not be located. Considerable effort was spent in the field trying to locate the benchmarks set during the Reserve studies (DWAF 1996, DWAF 2002) and/or the cross-section pins used for the Berg River Baseline Monitoring report (BRBM; Ractliffe *et al.* 2007) using a GPS. Of a possible 130 pins, only 12 were located but these were outermost pins and were insufficient to accurately guide a resurvey of the original cross-sections. Thus, a different approach was adopted for this chapter, which used aerial photographs and/or GoogleEarthPro© imagery, as discussed below. This was also not without its challenges, which included that most flight plans for aerial imagery only covered a portion of the basin, and the quality of many of the images was such that it was impossible to distinguish features, such as riparian zones and/or sand bars.

The initial plan to digitise channel and riparian proved impractical because the aerial images obtained from Department of Mapping and Surveys were not georeferenced, and to georeference them would have been expensive and time consuming, and thus unlikely to be something that would be done routinely for other river basins. In addition, it would have meant that all images, including GoogleEarth images would have had to be digitised. The experience with land-use (Chapter 3) also showed that even with considerable effort the additional quality and reliability of the data were insufficient to justify the time spent digitising, i.e., the value generated from digitising did not greatly exceed that of simpler methods. To overcome this constraint, a more rudimentary scaling of images was used. Aerial images were overlaid on the GoogleEarthPro© images and the scale was adjusted by matching up recognizable features in the landscape, such as roads, bridges and buildings, on both new and old images. This meant that the scale information imbedded in the GoogleEarthPro© images could be used to derive the semi-quantitative length and area data for the key characteristics listed above.

To decide on the periods and study reaches to be assessed, plans for historical aerial images were sourced from Department of Mapping and Surveys. The quality of the flightpath maps is fairly poor (Figure 5.1) and because the flight paths were designed to cover the basin (or parts thereof), they do not follow the river, but rather criss-cross the river. This means that images that cover the river needed to be sourced from different flightpaths, which made the process of identifying relevant images laborious and time-consuming. The process followed was: 1) for each period for which aerial impages were available, select flightpaths that touched the river; 2) check which, if any river reaches had imagery for multiple periods; 3) check the quality of the images available for those reaches; 4) discard images where the quality was too poor to make out the river channel or riparian zone.

From *c*. 2006 onwards, GoogleEarth imagery provides a full coverage of the basin. Prior to 2006, there were approximately 19 aerial surveys of the Western Cape; of those only one (1938) covered the Berg River macro-channel in its entirety, and six covered at least 50% of the macro-channel (Table 5.1). As flight plan 126 for year 1938 was the earliest survey and had a full coverage of the channel, images from the 1938 survey were used as reference images for describing change. The next available data after 1938 was flight plan 719, which was surveyed in 1977 with 85% coverage of the channel, but the images of 1978 were all too poor quality to be of any use, and were disgarded. A set of 16 reaches that had images for at least four periods including the earliest images (1938) were identified. A sub-set of two main river reaches in each of the three river zones was selected based on considerations of land-use, hydrological and invertebrate assessments done in Chapters 3, 4 and 6, respectively. The tributary reaches selected were those that had their confluence within or near the main river reaches selected. The result was the list of river reaches in Table 5.2.

5.2.1 Study reaches

The assessment focusses on five river reaches of the Berg River from its foothills to the head of the estuary. It also includes evaluation of the lower reaches of five tributaries at their confluence with the main Berg River (Table 5.2 and Figure 5.2). As discussed above, the selection of these study reaches was driven largely by the availability and quality of images that were available.

Each mainstem reach was approximately ~6 km long, and the tributary reaches extended from the confluence with the Berg River to ~2 km upstream of the confluence.

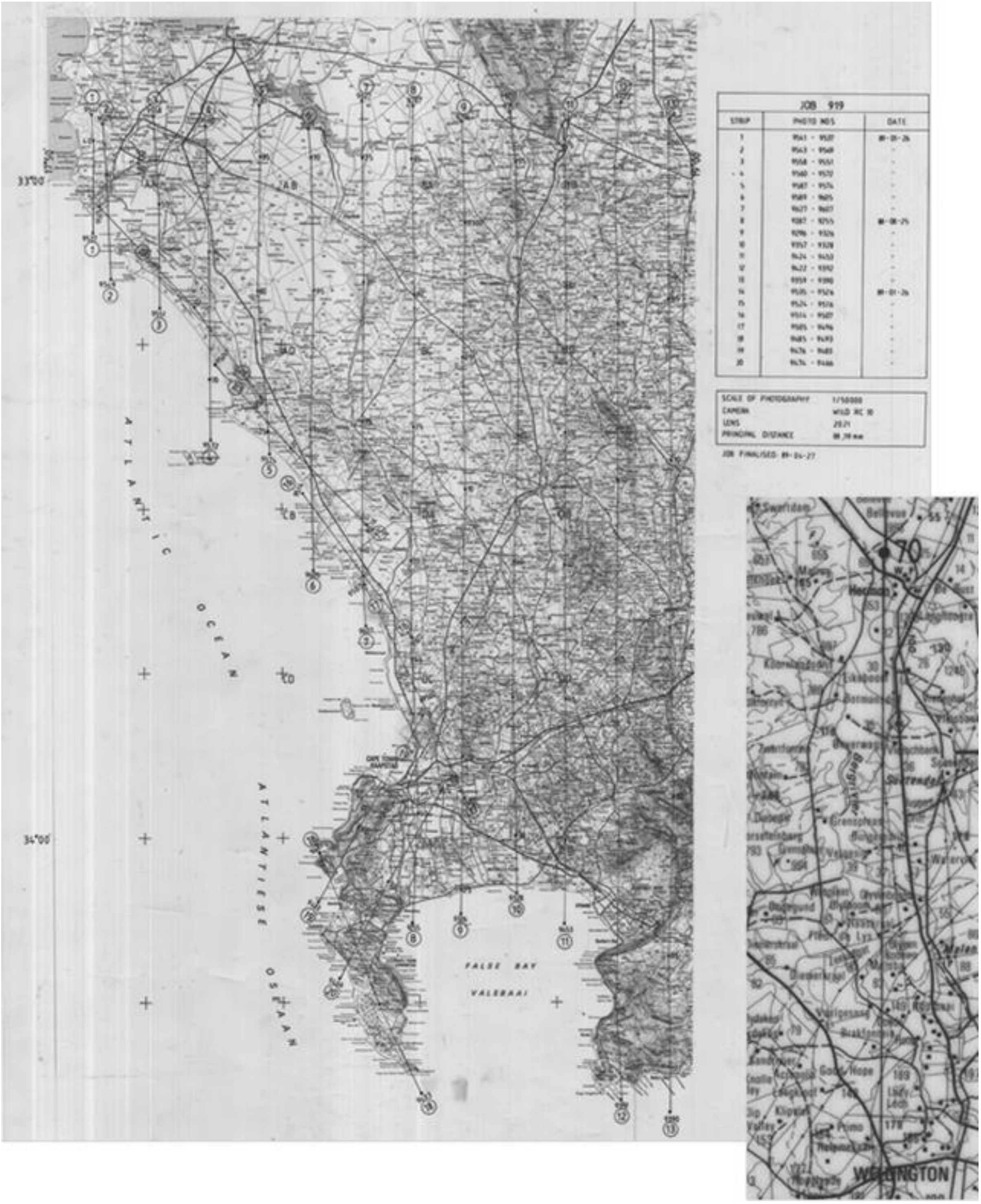

Figure 5.1 **Example of a flightpath index map from Department of Mapping and Surveys map with a section of the Berg River between Wellington and Hermon expanded. Flight paths are designated by the vertical lnes with numbered circles.**

Table 5.1 Aerial surveys that touched on the Berg River available from Department of Mapping and Surveys (Mowbray)

Flight plan No.	Section of the channel available	Channel % covered	Date
126	Source to mouth	100	1938
168	Kersefontein to mouth	3	1942
169	Only upstream of Franschoek	3	1942
225	Wellington to Hermon	4	1949
335	Franschhoek to Doljosaphat	10	1953
437	Die Brug to the mouth	20	1960
454	Twenty-fours River to downstream Misverstand	5	1960
675	Moravia (downstream Misverstand) to the mouth	20	1971
699	Paarl to Misverstand	50	1972
719	Wemmershoek to mouth	60	1973
786	Paarl to mouth	85	1977
911	Twenty-fours River to the mouth	75	1987
919	Soutkloof confluence to Kersefontein farm	5	1988/89
1033	Paarl to Misverstand	75	2000

Table 5.2 Location of study reaches where detailed data on channel shape were collected along the Berg River

Sub-basin	Reach number	Study reaches	Reach coordinates	
			Upstream end	Downstream end
Upper foothills	1	Berg River @ Franschhoek	-33.901640°; 19.053351°	-33.877784°; 19.033890°
	1a	Associated tributary: Franschhoek River	-33.890726°; 19.078846°	-33.881875°; 19.044107°
Lower foothills	2	Berg River @ Dwars	-33.841622°; 18.987581°	33.876636°; 19.025552°.
	2a	Associated tributary: Dwars River	-33.864403°; 18.985792°	-33.848988°; 18.993653°
	3	Berg River @ Hermon	-33.476677°; 18.938518°	-33.435000°; 18.956239°
	3a	Associated tributary: Doring River	-33.548111°; 18.907751°	-33.541475°; 18.926492°
Lowlands	4	Berg River @ Twenty-fours	-33.191918°; 18.934408°	-33.159081°; 18.899872°
	4a	Associated tributary: Twenty-fours River	-33.156382°; 18.972718°	-33.191749°; 18.935737°
	5	Berg River @ Misverstand	-33.014354°; 18.784943°	-32.972712°; 18.752218°
	5a	Associated tributary: Moorreesbergspruit River	-33.048239°; 18.789970°	-33.032967°; 18.789610°

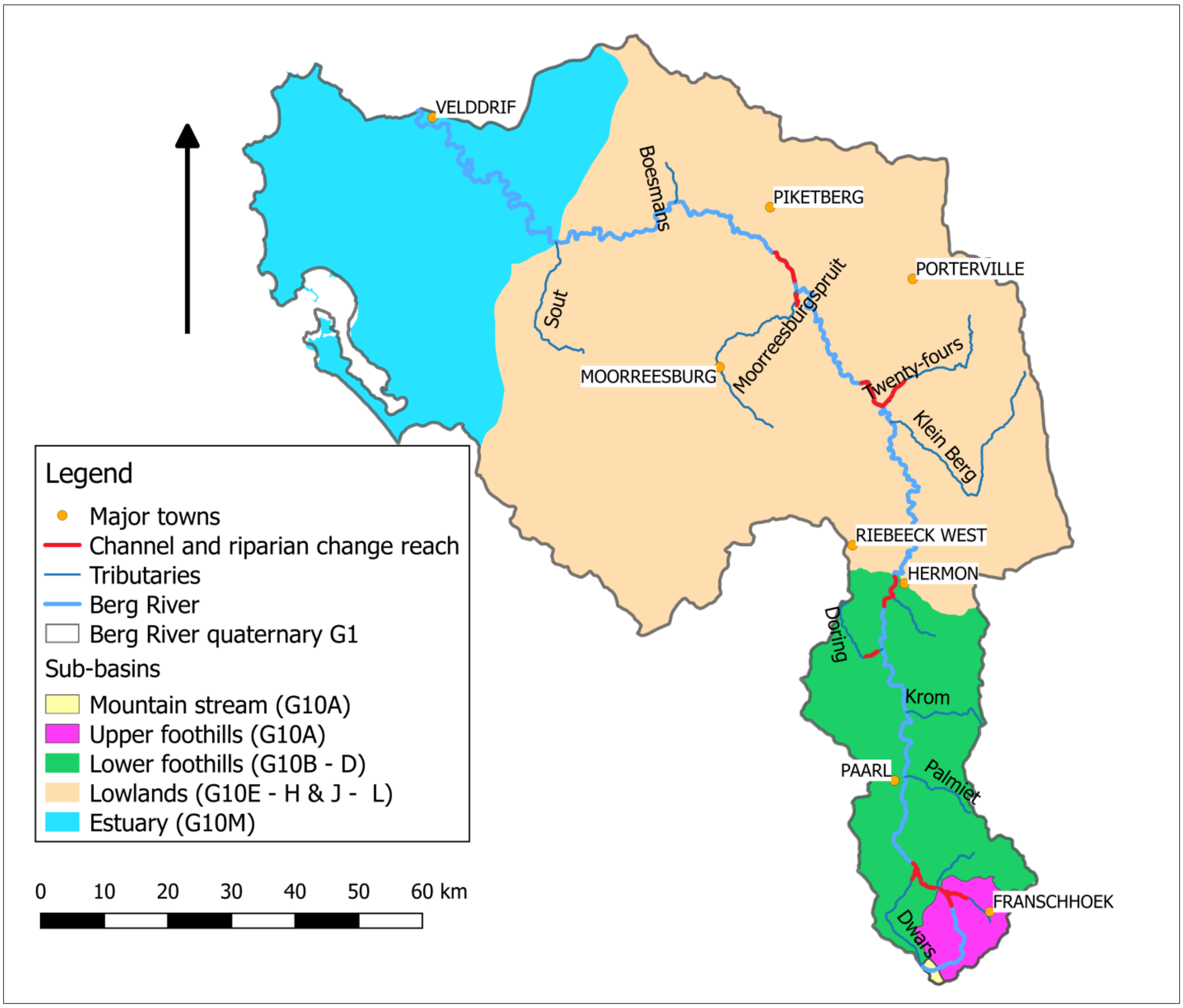

Figure 5.2 The location of the reaches (shown as red lines) where data on channel shape were collected on the Berg River and its tributaries

5.2.2 Periods assessed

The periods selected for analysis were 1938, 2003/06/09 and 2017 (Table 5.3). For the middle period, labelled 2003/06/09, there were no reaches where images taken on the same dates were available, and so a range of dates was used.

Table 5.3 Periods analysed for each reach

Sub-basin	Study reaches	Period		
Upper foothills	Berg River @ Franschhoek	1938	2003	2017
	Franschhoek River			
Lower foothills	Berg River @ Dwars	1938	2003	2017
	Dwars River			
	Berg River @ Hermon	1938	2009	2017
	Doring River			
Lowlands	Berg River @ Twenty-fours	1938	2006	2016/17
	Twenty-fours River			
	Berg River @ Misverstand			
	Moorreesbergspruit River			

5.2.3 Extraction of data from images

Channel characteristics and features at selected reaches were mapped from aerial photographs and/or GoogleEarthPro© imagery of the Berg River and its tributaries (e.g., Figure 5.3). Information on a predetermined set of channel features was then extracted. Some of this was semi-quantitative and some was qualitative. The term semi-quantitative is used because all of these measurements are more than likely affected to some extent by the sorts of problems linked to use of a disparate set of information highlighted by Hooke and Redmond (1989) and Uribelarrea *et al.* (2003) and discussed in the introduction.

Semi-qualitative information included description of channel form (based on Rosgen 1994) and estimates of continuity of the riparian zone.

- Channel form in terms of length of the thalweg, braiding, sinuosity/meandering,
- Extent of sandbanks or bars
- Extent of lateral floodplains
- Increases/decreases in the extent of channel and riparian areas ,
- Changes in riparian zone continuity (Gonzalez del Tanago and de Jalon 2006).

The semi-quantitative data were obtained from the available images by importing the scaled images into MSPowerPoint©, tracing the features on the scaled images, and then exporting the traced features into Excel. Actual measurements of area and length were captured from GoogleEarthPro© for the most recent time period 2017. The *.jpeg images for the traced shapes were entered into an Excel spreadsheet where a Visual Basic for Applications (VBA) macro script was written to produce areas/lengths of the traced features. The final area and length for all three periods were then calculated using measurements of area and length captured from GoogleEarthPro©. Macro-channel width; active channel width and riparian zone were combined into a single riparian zone/macro-channel width because it was not possible to distinguish between them with any level of precision or accuracy. The term 'riparian area' refers to areas directly adjacent to the active channel of a water course or waterbody that support vegetation communities, which are distinctly different to neighbouring terrestrial communities (Reinecke *et al.* 2007). For the purpose of the assessment, 'riparian vegetation' was assumed for areas that were not cultivated or otherwise completely altered from their natural state, e.g. through roads or revetments. The combination of channel and riparian vegetation has been referred to as 'channel and riparian area'. The extent and percentages of alien versus indigenous riparian vegetation could not be separated as the two could not be differentiated from the quality of images that were available for analysis. For the riparian vegetation, the following was recorded in terms of dominant and next abundant growth forms (trees, shrubs, reeds and grasses) in the riparian zone; and vegetation longitudinal continuity. Longitudinal continuity of the riparian zone was evaluated according to the percentage of the longitudinal axis of the riparian zone covered by riparian vegetation (Gonzalez del Tanago and de Jalon 2006) according to the following scale: (i) good = >75 % cover, (ii) fair = 25-75% cover and (iii) poor = <25% cover.

The qualitative estimates of channel shape and the percentage of each reach comprised of braided or single-thread channels were assessed visually for each reach. River braiding is characterized by channel division around alluvial islands (Leopold and Wolman 1957).

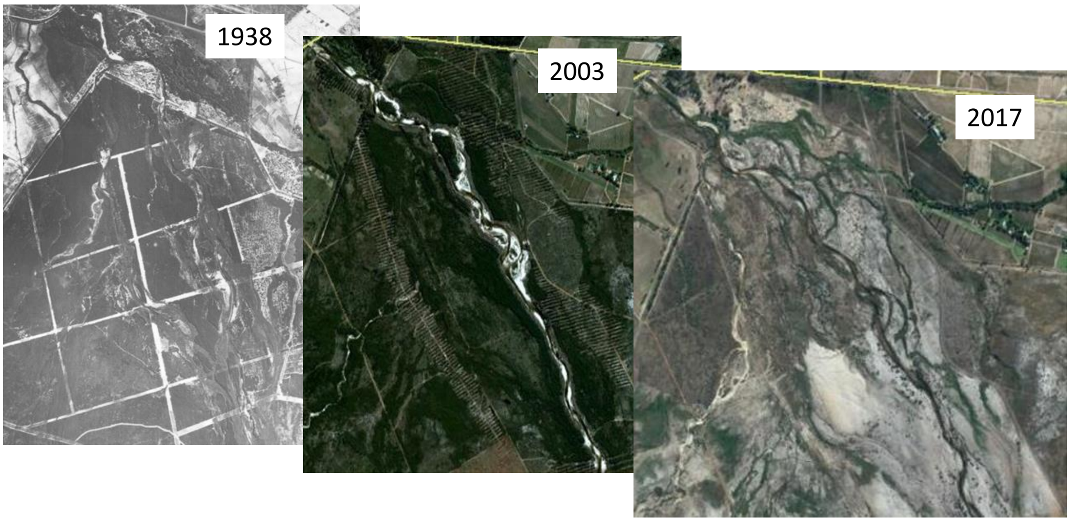

Figure 5.3 Images used for Reach 1: Berg River at Franschhoek

Change in channel form of the entire reach was calculated over time using an index of sinuosity. Sinuosity was calculated as the ratio of thalweg length to the straight-line valley length (e.g. Schumm 1963). A reach with sinuosity ≥1.5 is considered meandering (Leopold and Wolman 1957).

5.2.4 Data analyses

The proportions and area data were analysed using the traced images, and summary tables and graphs produced in MSExcel©. To highlight changes in proportions of the measured and described variables, these were evaluated for each reach based on three dates: 1938, 2003/6 and 2017. The extent to which the recorded changes mirror other changes in the catchment is also discussed in this chapter.

The data were also carried forward to the meta-analyses in Chapter 7 where they were compared against results from the land-use (Chapter 3) and flow regime (Chapter 4) assessments in an effort to identify possible drivers of change.

5.3 Results

The traced images for the study reaches in each period listed in Table 5.2 are presented for the main Berg River in Figure 5.4 and for the tributaries in Figure 5.5. These comprise tracings of: the thalweg of the mainstem and each location; in-channel sand bars and islands; the floodplain, and; channel and riparian area.

The data extracted for the main Berg River and using these images are presented in Table 5.4 and Table 5.5, respectively. These tables show, for each reach, the tracings for the three periods analysed. An X means that the feature did not exist, e.g., sand banks and bars at Reach 3 2009 and 2017.

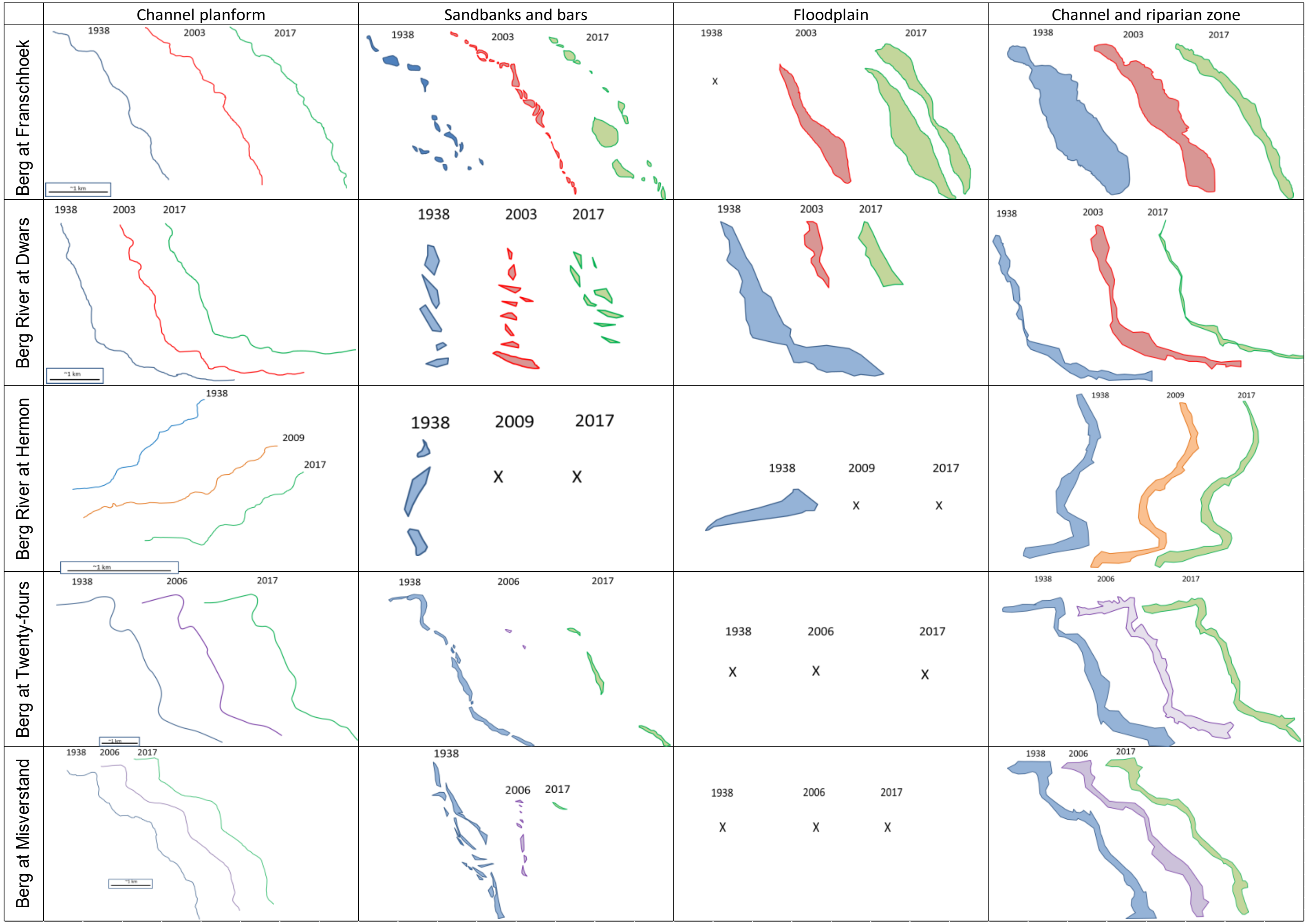

Figure 5.4 Traced images of channel form, sand banks and bars, floodplain and riparian area for the main Berg River reaches

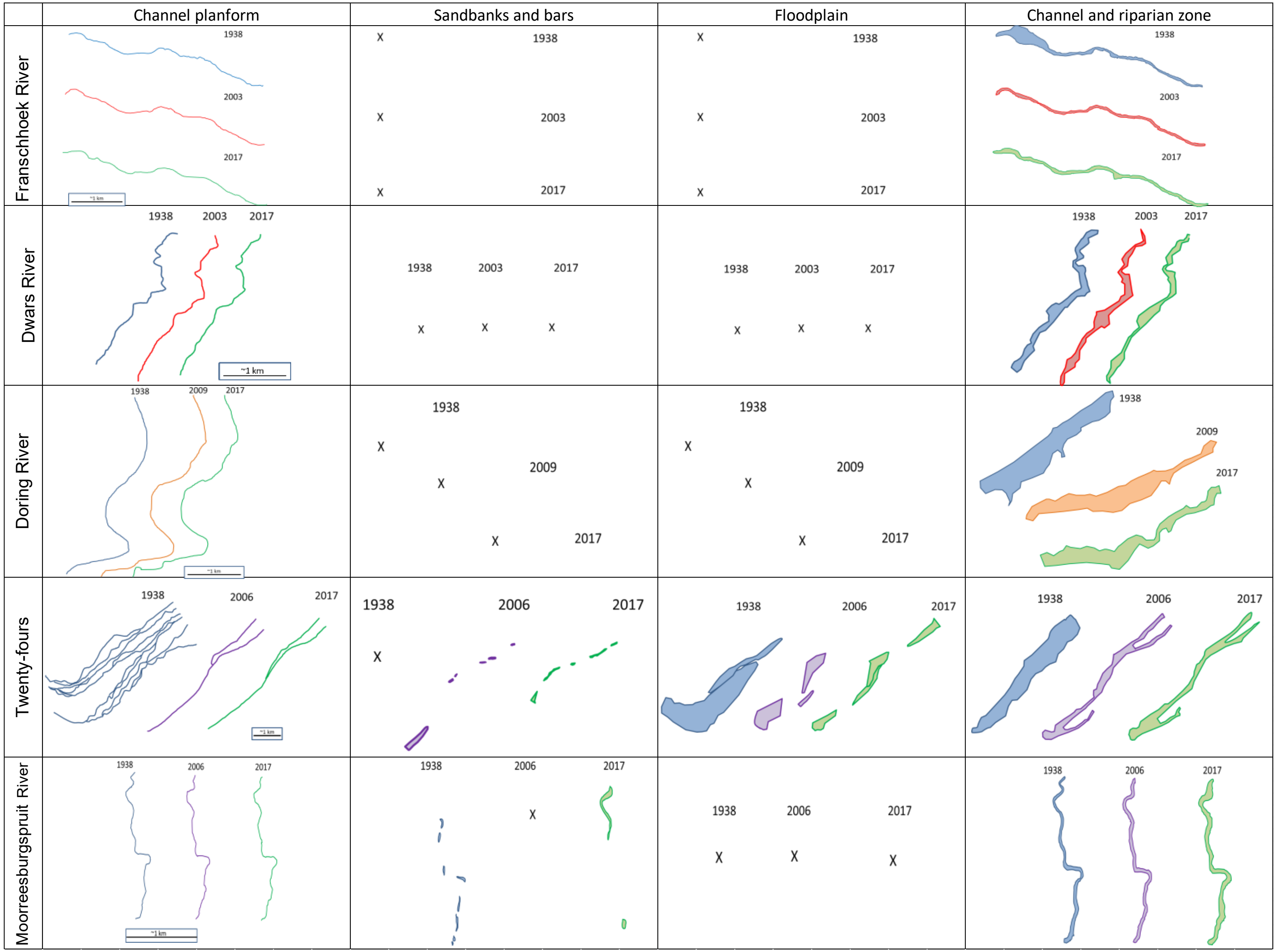

Figure 5.5 Traced images of channel form, sand banks and bars, floodplain and riparian area for the tributary reaches

Table 5.4 The data extracted from the aerial images for the reaches on the Berg River

Reach #	Units	1			2			3			4			5		
Aspect		Berg River @ Franschhoek			Berg River @ Dwars			Berg River @ Hermon			Berg River @ Twenty-fours			Berg River @ Misverstand		
		1938	2003	2017	1938	2003	2017	1938	2003	2017	1938	2006	2017	1938	2006	2017
Length of the thalweg	m	3646.2	2913.9	3596.0	7093.8	6393.1	6033.0	5978.4	6813.3	6168	6257.7	5672.3	6006	5602.9	5302.5	5748.9
Sinuosity	Index	1.61	1.25	1.51	1.78	1.59	1.54	1.95	2.09	1.87	1.29	1.30	1.29	1.32	1.35	1.42
Channel and riparian area	km^2	1.12	0.73	0.37	0.77	1.04	0.26	1.05	0.86	0.79	2.18	1.69	1.34	0.84	0.77	0.60
Sandbanks or bars	km^2	0.14	0.14	0.28	0.09	0.07	0.05	0.09	0	0	0.09	01	0.02	0.09	0.03	0.01
Floodplain	km^2	0	1.14	1.47	1.21	0.17	0.26	0.31	0.30	0	0	0	0	0	0	0
Braiding	%	90	20	60	50	10	15	50	5	5	5	0	0	90	40	20
Riparian continuity	%	>75	>75	>75	25-75	25-75	<25	25-75	>75	25-75	25-75	25-75	25-75	25-75	25-75	25-75
Trees and shrubs	%	50	40	20	50	65	5	5	90	10	40	70	70	5	30	40
Reeds and grasses	%	50	60	80	50	35	95	95	10	90	60	30	30	95	70	60

Table 5.5 The data extracted from the aerial images for the tributary reaches

Reach #	Units	1a			2a			3a			4a			5a		
Aspect		Franschhoek River			Dwars River			Doring River			Twenty-fours River			Moorreesbergspruit		
		1938	2003	2017	1938	2003	2017	1938	2003	2017	1938	2006	2017	1938	2006	2017
Length of the thalweg	m	3361.4	3221.8	3417	2045	2005	2003	1739.4	1790	1996	5371.6	4867	5337	1942	2128	2031
Sinuosity	Index	1.42	1.33	1.38	1.77	1.69	1.67	1.46	1.42	1.66	1.34	1.13	1.20	1.49	1.65	1.58
Channel and riparian area	km^2	0.14	0.07	0.10	0.29	0.25	0.14	0.40	0.35	0.29	1.83	0.80	0.93	0.06	0.06	0.09
Sandbanks or bars	km^2	0	0	0	0	0	0	0	0	0	0	01	0.01	0.01	0	0.01
Floodplain	km^2	0	0	0	0	0	0	0	0	0	1.63	0.71	0.81	0	0	0
Braiding	%	0	0	0	20	20	20	0	0	0	85	30	20	0	0	0
Riparian continuity	%	>75	>75	>75	>75	>75	25-75	25-75	25-75	25-75	25-75	25-75	25-75	25-75	25-75	25-75
Trees and shrubs	%	80	60	60	90	90	70	70	50	60	60	40	30	20	20	60
Reeds and grasses	%	20	40	40	10	10	30	30	50	40	40	60	70	80	80	40

In general, and with the exception of Reach 1a, the data for the reaches on the Berg River show a progressive decline in braiding and a loss of woody riparian vegetation (Table 5.4). Channel and riparian area (Figure 5.6), braiding and sinuousity (Figure 5.7), declined markedly between 1938 and 2003/6 but increased again in 2017. Sand banks and bars tended to reduce in size and increase in number between 1938 and 2017; before disappearing completely in some cases e.g. Berg at Misverstand. Reach 1a and 3a shows an increased braiding over time, particularly between 2003/6 and 2017. The data for the tributaries (Table 5.5) showed similar trends to those for the main channel in that, for the most part, they have become progressively smaller and less braided. The results for individual reaches are discussed below.

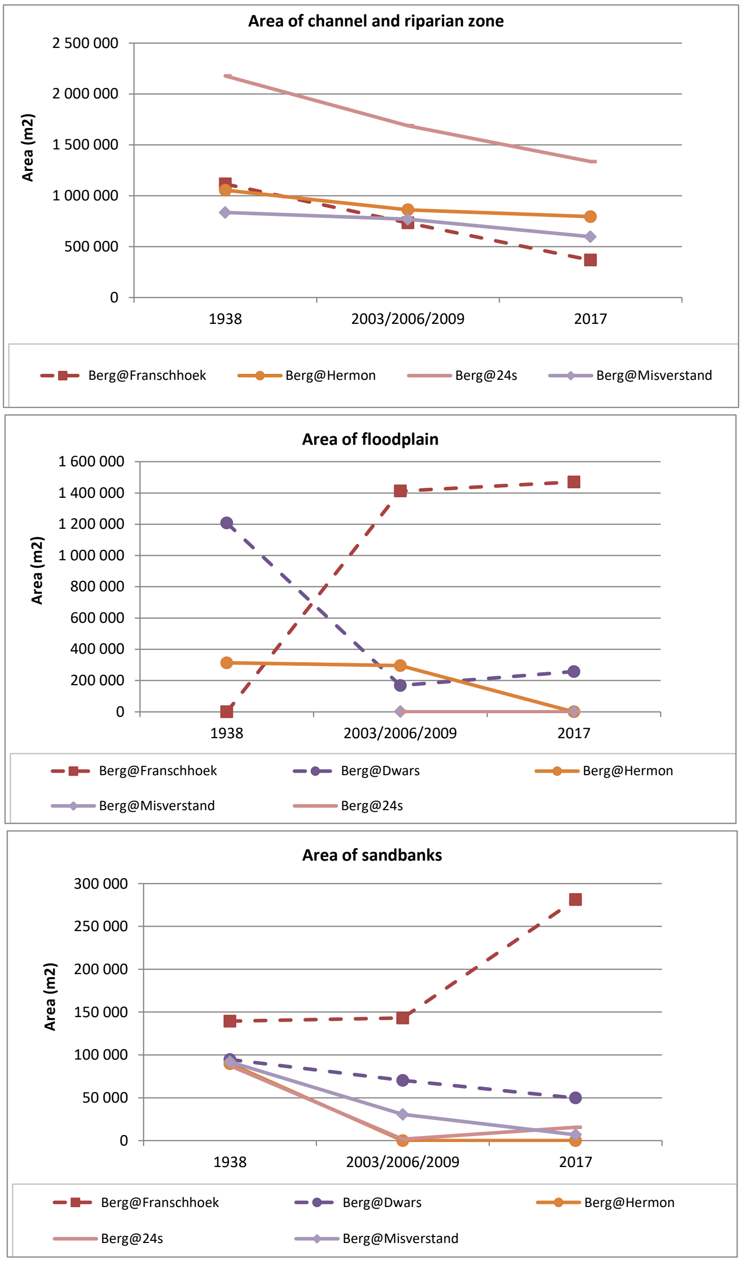

Figure 5.6 Channel and riparian, floodplain and sandbank area for the main Berg River reaches for 1938, 2003/6 and 2017

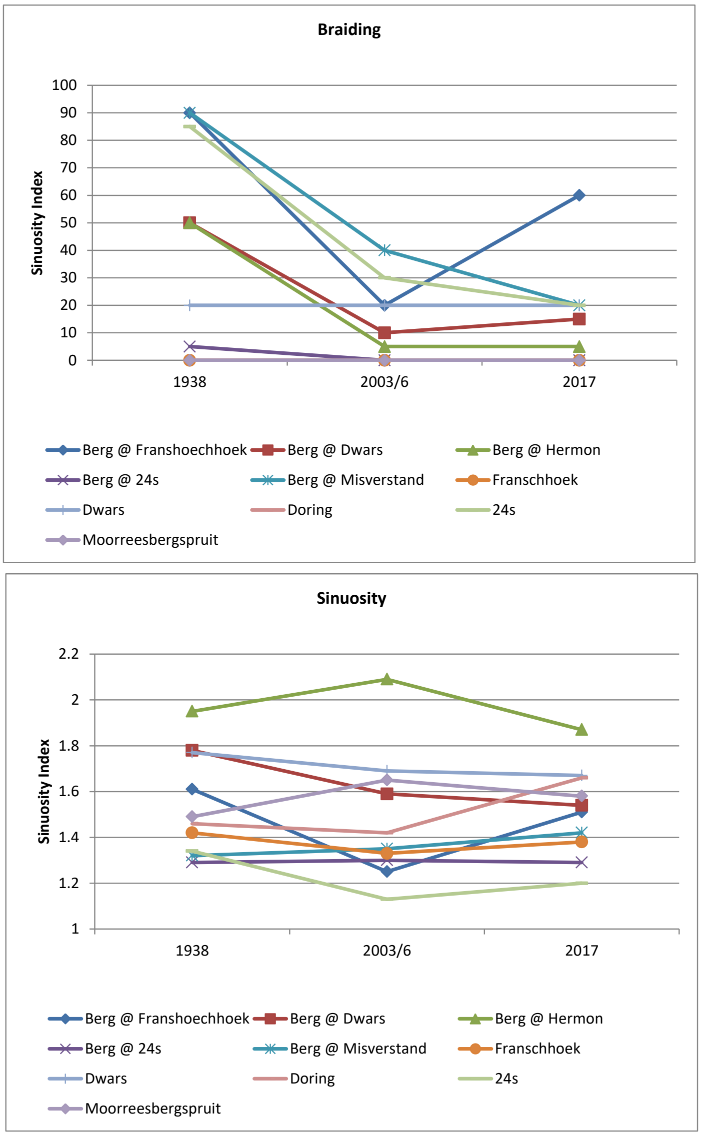

Figure 5.7 Braiding and sinuosity for all study reaches for 1938, 2003/06 and 2017

5.3.1 Channel changes in the Berg River

5.3.1.1 Upper foothills

The upper foothills of the Berg River are represented by Reach 1: Berg River at Franschhoek, which is a 6-km reach from downstream of the Berg River Dam to the confluence with the Franschoek River. In 1938, this reach was heavily braided (~90% of the reach; Figure 5.4; Table 5.4), but by 2003, the braided sections had been reduced to ~20% of the reach. By 2017, following removal of the surrounding forestry and rehabilitation, ~60% of the reach was braided. Sinuousity followed a similar trend, with the greatest sinuosity recorded in 1938, and the lowest in 2003, before restoration (Figure 5.3; Table 5.4).

The area covered by the channel and riparian zone combined have declined from 1938 to 2017, and was narrowest post the closure of the Berg River Dam (Figure 5.6). This was offset by a general increase in floodplain size over time, with a change from a single floodplain in 2003 (left bank) to floodplains on both banks in 2017 (Figure 5.4; Figure 5.6; Table 5.4).

Sandbanks and bars were present in the channel in all three periods, but differed in size, shape and location over time, with a general trend towards fewer larger sandbanks over time (Figure 5.4). In 2017, the size of sandbars was double relative to 2003.

The images show that, in 1938, this reach was flanked by a thick, continuous band of riparian vegetation (>75% continuity), dominated by trees and shrubs (80%; Figure 5.4; Table 5.4). The riparian vegetation in 2003 was still >75% continuous, but the trees and shrubs made up only 40% of the community and had been largely replaced by grass/reeds (60%). By 2017, following removal of the surrounding forestry and rehabilitation, continuity of the riparian vegetation was still >75%, but woody vegetation (trees and shrubs) comprised only ~20% of the community, and grass/reeds dominated at ~80%.

5.3.1.2 Lower foothills:

The lower foothills of the Berg River are represented by two reaches: Reach 2: Berg River at Dwars and Reach 3: Berg River at Hermon.

Reach 2: Berg River at Dwars is a 6-km reach located between the confluence with the Wemmershoek River to the confluence with the Dwars River. In 1938, there were multiple side channels on the right bank of the active channel. By 2003, these had disappeared under cultivation and the river was a single thread channel. In 2017, one or two of these old channels were again evident, possibly as a result of flooding in the intervening years. The extent of the channel and riparian zone was 0.77 km^2 in 1938, 1.04 km^2 in 2003 and 0.26 km^2 in 2017 (Table 5.4; Figure 5.6). The river channel was more sinuous in 1938 and straighter in 2017. There were fewer sandbanks and bars in 1938 but those that were present were large. Sandbanks became progresively more numerous and smaller from 1938 to 2003 to 2017. The floodplain also reduced in extent between 1938 (1.21 km^2) and 2003 (0.17 km^2), and increased again slightly in 2017 (0.26 km^2; Figure 5.4; Figure 5.6; Table 5.4). Riparian vegetation continuity was fair >75% in 1938 and 2003, but reduced to 25-75% in 2017 (Table 5.4).

Reach 3: Berg River at Hermon is the 6-km stretch upstream of the R46 Hermon Road Bridge. In 1938, ~50% of the channel was braided, but this was not evident in 2003 or 2017, when only ~5% of the channel was braided. Sinuosity was fairly stable across the periods assessed (Table 5.4), as was riparian vegetation continuity, although this reach differed from the two upstream reaches, in that in 1938 trees made up only 5% of the riparian vegetation, but in 2003 they made up 90%. By 2017, this has changed again, and trees comprised 10% of the riparian vegetation community.

5.3.1.3 Lowlands

The lowlands of the Berg River are represented by two reaches: Reach 4: Berg River at Twenty-fours and Reach 5: Berg River at Misverstand.

Reach 4: Berg River at Twenty-fours is a 6-km stretch downstream of the Twenty-fours River confluence with the Berg River. In 1938, ~5% of the reach was braided but this was removed between then and 2006, and it was a single change in 2006 and 2017. The sinuosity of the channel remained the same over time, although the extent of the channel and riparian areas decreased over time (Table 5.4; Figure 5.6): in 1938 it was 1.29 km^2, in 2006 it was 1.69 km^2 and in 2017 it was 1.34 km^2 (Table 5.4; Figure 5.4). Sandbanks and bars were plentiful and large in 1938 and were very much reduced in size and number by 2006 with a few having redeveloped by 2017 (Figure 5.4; Figure 5.6). Between 1938 and 2017, ~82% (0.09 km^2) of sandbanks and bars were lost. No floodplain was present here. The nature of the riparian vegetation changed from a community dominated by reeds and grasses with ~50% continuity in 1938, to one dominated by (70%) with increased continuity >70% in 2006 and 2017.

Reach 5: Berg River at Misverstand is a 6-km stretch downstream of Misverstand Dam. Ninety percent of channel was heavily braided in 1938, but this reduced over time, such that 40% of the channel was braided in 2006 and only 20% in 2017; although sinuousity was similar throughout (Table 5.4). Extent of the channel and riparian area decreased over time, with an 8% reduction between 1938 and 2006, followed by further 22% reduction between 2006 and 2017 (Table 5.4; Figure 5.6). The size of sandbars also declined over time (Table 5.4; Figure 5.6). The composition of the riparian vegetation also did not change, and was dominated by reeds and grasses throughout, although the percentage of trees increased slightly over time (Table 5.4). The riparian vegetation continuity remained was 25-75% in all the periods.

5.3.2 Channel changes in the tributaries

5.3.2.1 Franschhoek River

Reach 1a (Franschhoek River) is a 2-km reach on the Franschhoek River immediately upstream from its confluence with the Berg River. There were no obvious changes in river planform evident in this tributary over the period assessed, apart from the fact that the channel was more sinuous in 1938 and 2017 respectively and straighter in 2003. Also, the area of riparian zone, which was thick near the confluence, reduced over time (Figure 5.5). Continuity of the riparian vegetation did not change over tme – but in 1938, the Franschhoek River was a single thread channel surrounded by a thick continuous (continuity >75%) riparian vegetation of about 80% trees and 20% shrubs (Table 5.5). In 2003 the extent of shrubs was about twice that of 1938, and the ratio of trees to shrubs was about 60:40 (Table 5.5). This remained unchanged from 2003 and 2017.

5.3.2.2 Dwars River

Reach 2a (Dwars River) is a 2-km reach on the Dwars River immediately upstream from its confluence with the Berg River. The channel was single thread, with poor riparian vegetation connectivity (<25%) across all periods. Although the percentage of reeds and grasses increased over time, trees remained the most dominant type of vegetation along this reach. Sandbanks and floodplains were absent from all periods. The extent of channel and riparian vegetation was largest in 1938 and smallest in 2017, 0.29 km^2 and 0.14 km^2 respectively, with more than half of the area lost between the two periods (Figure 5.5; Table 5.5). There was a decrease in the area of the channel and riparian zone, 15% was lost between 1938 and 2003 and a further 43% was lost by 2017 (Table 5.5).

5.3.2.3 Doring River

Reach 3a (Doring River) is a 2-km stretch on the Doring River immediately upstream from its confluence with the Berg River. The channel remained single thread over time, but was more sinuous in 1938; and straighter in 2017 (Figure 5.5; Table 5.5). The extent of the channel and riparian area decreased slightly over time, but channel shape was similar and there were no sand banks or floodplains in any of the periods assessed (Table 5.5). Riparian continuity was fair (25-75%) in all periods (Table 5.5), and the composition was a fairly even mixture of woody trees and shrubs, and reeds and grasses (Table 5.5).

5.3.2.4 Twenty-fours River

Reach 4a (Twenty-fours River) is a 2-km stretch on the Twenty-fours River immediately upstream from its confluence with the Berg River. In 1938 more than 85% of the channel was braided, there were more than 13 river channels. By 2006, this braiding had been radically reduced to less than 30% of the channel with braiding, which comprised just one main and one side channel by 2006. There was a further reduction in the braiding by 2017, with only 20% of the channel showing any sign of braiding (Table 5.5). The sinuousity of the reach was greatest in 1938 and least in 2006 (Table 5.5). There were no sandbanks in 1938, but these had developed by 2006 and had increased by 2017 (Figure 5.5; Table 5.5). The floodplain size was largest in 1938 and smallest in 2006. Continuity of the riparian vegetation remained fair (25-75%) throughout the periods assessed, but this changed from a community dominated by trees in 1938 to one dominated by grasses and reeds in 2017 (Figure 5.5; Table 5.5).

5.3.2.5 Moorreesbergspruit

Reach 5a (Moorreesburgspruit) is a 2-km stretch on the Moorreesburgspruit immediately upstream from its confluence with the Berg River. The reach was single-thread over all periods, and although the channel became more sinuous in 2006 and 2017 having been straighter in 1938 (Figure 5.5; Table 5.5), the changes were small. There was little or no change in channel and riparian area over time and there was no floodplain in any of the periods (Table 5.5). Sandbanks and bars reduced considerably between 1938 and 2006, but one or two large sand bars appeared again in 2017 (Figure 5.5). The riparian vegetation continuity was fair (25-75%) over all periods. Reeds and grasses were dominant (80%) in

1938 and 2006, but in 2017 trees made up 60% of the riparian vegetation community (Table 5.5).

5.4 Discussion

Human influences have substantially changed the natural flow of rivers around the world, disrupting the dynamic equilibrium between the movement of both water and sediment (Poff *et al.* 1997) with cascading effects on the physical and ecological integrity of rivers including the riparian area (Stromberg *et al.* 1996; del Tanago and de Jalon 2006). Downstream effects of development include altered river discharge, decreased suspended sediment, channel incision, flooding, floodplain and channel narrowing, stream meandering, and a decrease in diversity of the riparian habitat; accompanying such changes are shifts in riparian plant community composition (Nilsson *et al.* 1991; Stromberg 1993; Busch and Smith 1995; Gilvear *et al.* 2002). The extent of manipulations is expected to continue to increase with growth in human population and demand for energy, irrigation and industrial activities (Poff *et al.* 1997). For example in Japan floodplains are of strategic importance, they cover less than 10% of land but contain 50% of the population and more than 70% of the assets are located on 'former' floodplains (Nakamura *et al*. 2006).

Given the extensive land use changes that had already taken place by 1938 (Chapter 3; Burman and Levin 1979), it is likely that the planform and riparian vegetation of rivers in the Berg River Basin were already changed from their natural state by then. Nonetheless, as hypothesized, the changes in channel planform, habitat diversity and riparian vegetation between then and now (2017) have been extensive. In general, as expected, these changes tend towards narrower systems with less habitat diversity and less protection from the surrounding activities; there was a progressive decline in channel braiding and loss of woody riparian vegetation, and for the most part sandbars and riparian floodplains present in 1938 were lost by 2003. In the Likangala River catchment in Malawi, removal of vegetation led to increased sediments in the river and an overall increase in agricultural lands (Jamu *et al*. 2003).

There are some obvious exceptions to this. Changes in the Berg River at Franschhoek followed the expected trends between 1938 and 2003, but these were dramatically reversed by 2017. This reversal was a result of an extensive investment in rehabilitation of this river reach following the removal of the surrounding forestry and construction of the Berg River Dam, which involved alien clearing, revegetating with indigenous plants and construction of a continuous gabion revetment (Guillaume Nel Environmental Consultants 2017). Following rehabilitation, braiding and sinuosity inceased and the area covered by the channel and its riparian vegetation increased, although the vegetation remained dominated by reeds and grasses. Another exeption was Reach 5a (Moorreesburgspruit), which although single thread over all periods became slightly more sinuous, sandbanks came and went between periods and the riparian vegetation community shifted from one dominated by reeds and grasses to one dominated by woody vegetation. The reasons for Moorreesburgspruit's deviation from the more common trends are not immediately clear.

In several cases, the change in composition of the riparian vegetation appears to be linked to invasion of the riparian areas by woody exotics, such as *Eucalyptus* spp *and Acacia* spp. According to Bunn and Arthington (2002) the changes in or loss of wet-dry cycles favours exotic species because they are more resilient than the indigenous species. The presence of these trees in riparian areas of the Western Cape were recognised as a major problem in the latter part of the 20th Century (e.g., Macdonald and Richardson 1986; Richardson *et al.* 1989; Le Maitre *et al.* 2002; Richardson *et al.* 2007), which resulted in the Working for Water (WfW) programme. WfW programme was launched in 1995 and is managed by the Department of Environmental Affairs. The programme is aimed at removing invasive alien plants from river catchments, and has resulted in the removal of thousands of hectares of alien trees from the riparian areas of rivers in the Western Cape. At the time of its launch, the Berg and Breede basins were deemed to be the most heavily invaded in South Africa (Le Maitre *et al.* 2000) and extensive clearing took place along the Berg River between *c*. 2000 and 2017. The results of the clearing are clearly visible in the changes reported here, and in several of the 2003/2006 the dominance of woody vegetation is more than likely due to the alien invasion and reduction shown in 2017 is a result of alien clearing by WfW.

There are several other changes that may have arisen as a result of WfW activities. Chief among these is that the invasion of alien vegetation, and its subsequent clearing, appears to have reduced all woody vegetation, not just alien species. Also, the removal of the woody vegetation is linked with a further decline in channel and riparian area. In other words, although removal of exotic vegetation undoubtedly has benefits for water supply and ecosystem functions (Richardson *et al.* 1989; Richardson *et al.* 2007), the results of this study suggest that removal of the alien vegetation reduced the buffer zone around the rivers in the Berg River Basin, which is known to have negative knock-on effects on habitat quality (e.g., Norris 1993; Allan 1995; Roth *et al.* 1996) and diversity (Bennett and Mulongoy 2006). Indeed these systems are core habitat for many semi-aquatic and terrestrial “ecotone” species (Semlitsch and Jensen 2001). There are reports that WfW teams cleared away the indigenous trees as well as alien trees, as clearing was done prior to the teams being trained to recognize alien and indigenous species. For instance, on the Molenaars River, downstream of the Du Toitskloof Tunnel, large Afromontane trees along the river, which were surrounded by exotic *Acacia mearnsii*, were cut down by WfW (Dr Karl Reinecke, Southern Waters, pers. comm.). There are similar reports downstream of the Berg River Dam.

The second hypothesis that different land-uses affect the riparian area and river channel structure in different ways was not supported, at least outside of the urban areas. Certainly at the level of analysis that was feasible for this study, it was not possible to distinguish between the effects of say viticuluture and heat farming. Some tributaries changed far more than others. For example, the manipulation of the Twenty-fours River was extensive and dried-up river channels have been taken over for viticulture. However, it is not possible generally to link extent or type of change to any particular land-use. Partly this was because the changes were similar at all sites and because in the Berg River differences in land-use are more than likely masked by differences in position in the catchment as forestry and viticulture tends to occur in the upper reaches and wheat farming in the lower reaches (Chapter 3).

The links between changes in channel morphology and knock-on effects on hydrology are well documented in the scientific literature (e.g. Lane 1955; Schumm 1969; Gregory 1977; Clark and Wilcock 2000). Of these, possibly the most recognised are the changes in flood hydrographs as a result of hardening of a catchment (Booth 1990; Leopold 1972; Gregory 2006; Lauer *et al.* 2017). It can be argued that the loss of channel area and complexity, the narrowing of the riparian zone and loss of woody vegetation, and the loss of sandbars and floodplains represent a type of catchment hardening. A sinuous or braided channel, for instance, slows the flow of water more than does a straight and/or single channel (Leopold *et al.* 1960). It is relevant therefore, that Chapter 4 reported a reduction in the magnitude and duration of mid-sized floods, particularly in the lower parts of the river.

Channel planform and riparian vegetation are particularly sensitive to development and serve as good indicators of environmental change (Booth 1990). However, it is not always easy to record and analyse these changes. In this study, attempts to rely on previously surveyed cross-sections were unsuccessful as bench marks could not be located despite considerable efforts in the field. With a better understanding of the extent of change in channel planform gained from analysis of historic aerial photographs, it is unsurprising that the survey pins could not be found. The difficulty in locating previously surveyed cross-sections is not unique to this study. Leopold (1973) reported that when surveying the Watts Branch in Washington D.C from 1952-1972, it was relatively easy to locate the survey pins in the first few years but in the later years the task became more difficult particularly where the land had been transformed by farming and/or urban development. Pasture land became grassland then in later years huge amounts of silt were deposited into the channel and banks due to upstream developments and the digging required to find each pin was enormous. Added to this lateral river migration exposed and destroyed some pins (Leopold 1973). This took place for benchmarks whose position had been recorded 12 months earlier. In the case of the Berg River, some of the benchmarks sought had last been seen 10-15 years earlier.

It was, however, possible to track changes in channel morphology and riparian zone vegetation from historical aerial images. Although it is recognised and acknowledged that the use of aerial images is not without it problems, the results presented clearly show that channels of both the Berg River and its tributaries have become narrower and less diverse as a result of development in the basin. They also show that with a concerted effort it is possible to reverse some of the more immediate changes, e.g., Reach 1 – Berg River at Franschhoek, which would undoubtedly have knock-on benefits for biodiversity, but also for water management and supply, throughout the basin.

6 Historical changes in aquatic macroinvertebrate communities in the Berg River

6.1 Introduction

The main aim to this chapter is to examine the status of macroinvertebrate communities of the Berg River and evaluate the changes over time by comparing their types and abundance. This will give an insight to the different groups of invertebrates that were available then and those that are recently present along the length of the river, from upper foothills to the lowlands.

River basin features, including climate, slope, altitude, geology and land-use, influence river flow and the structure and availability of habitat for biota (Hynes 1975; O'Keeffe *et al.* 1989; Allanson *et al.* 1990; Dallas and Day 1993; Richards *et al.* 1996; Davies and Day 1998; Ractliffe *et al.* 2007). In the south-western Cape, which is known for its high degree of endemism (Harrison and Agnew 1962; Wishart and Day 2002; Dallas and Day 2007), biotic communities have been shown to be distinct at a basin level (King and Schael 2001; Wishart and Day 2002) and display what are referred to as 'catchment signatures' (Schael 2005).

Altitude has also been identified as a factor that affects structuring of different faunal compositions (Marchant *et al.* 1995; Carter *et al.* 1996; and Dallas 2004) showed that macroinvertebrate assemblages in the Western Cape differ down the length in line with the predictions of the River Continuum Concept (RCC, Vannote *et al.* 1980) and modifications thereto (e.g. Minshall *et al.* 1985). The RCC considers rivers as ecosystems where processes in downstream areas are linked to those in the upstream reaches by the unidirectional flow of water and materials (Naiman *et al.* 1988). The theory is that biological assemblages are organised on the basis of the size and availability of organic matter, and because this gradually becomes finer down the length of a river, the biotic communities, such as macroinvertebrates, will change down the length of the river (Vannote *et al.* 1980; Walker 1985), although there is some debate as to whether the details described in the RCC apply to all river types (Winterbourn *et al.* 1981; O'Keeffe *et al.* 1989; Gale 1992). There are also other factors controlling distribution and the composition of macroinvertebrate communities are also determined by the hydraulic conditions in the different river reaches. In mountain streams, where natural disturbance is high and unpredictable (Townsend 1989) and the landscape is erosional, species making up the communities have different life-history strategies from those in the lower reaches where disturbance is lower and the landscape is depositional. Vannote *et al.* (1980) suggest that increasing habitat heterogeneity promotes faunal diversity, while increasing physical stability dampens faunal diversity, although again there is some debate about this, and Winterbourn *et al.* (1981) suggest that increased physical stability combined with increased heterogeneity enhances faunal diversity. What is undisputed is that, within the river continuum, complex interactions between flow and habitat control the distribution, abundance and diversity of riverine biota (Poff and Allan 1995; Ward *et al.* 1999; Nilson and Sverdmark 2002), and that for any given basin, these are controlled by the flow regime (Poff *et al.* 1997; Richter *et al.* 2003; Naiman *et al.* 2008). Thus, the temporal and spatial variability of macroinvertebrate communities are influenced by a wide range of factors including landscape, physical, chemical (water quality), aquatic and riparian vegetation, river flow together with other hydraulic parameters, river morphology (slope), sediment load, habitat/ biotope availability and stability and regional climatic seasonality (rainfall) among others (Gore 1978; Richards *et al.* 1993; Bispo *et al.* 2006; Haidekker and Hering 2007).

For disturbed systems, the RCC was extended into the Serial Discontinuity Concept (SDC; Ward and Stanford 1983), which holds that the RCC is conceptually sound, but that river ecosystems are seldom, if ever, uninterrupted continua. The SDC predicts that interruption will cause discontinuity in the stream continuum (Ward and Stanford 1983; Minshall *et al.* 1985), which may or may not recover to natural levels with distance downstream, or removal, of the disturbance. These interruptions can take the form of alterations in, e.g., flow, sediment supply, channel shape, temperature (e.g., Tate and Heiny 1995; McIntosh *et al.* 2002; Arthington *et al.* 2004; Dallas and Rivers-Moore 2012; Nautiyal 2013). Impoundments also impair the natural continuum and thus may disrupt life stage developments and diversity of biota (Nautiyal 2013). Transfer of water between river basins is also a major distruptor (Snaddon *et al.* 2000) and the transfer of water from Theewaterskloof Dam of the Riviersonderend Basin into the upper Berg River resulted in decreased taxon richness, loss of sensitive taxa and an increase in collectors-predators below the outlet (Snaddon and Davies 1998).

The impacts of various human activities on rivers have rendered them among the most threatened ecosystems on Earth (Nel *et al.* 2009; Speed 2016). This is because of the longitudinal linkages discussed above but also because the river's ecological integrity is a reflection of all the activities in the lotic landscape that it drains (Hynes 1975; O'Keeffe *et al.* 1989; Allanson *et al.* 1990; Dallas and Day 1993; Davies and Day 1998). The river therefore bears the consequences not only of abstraction and pollution (Masase *et al.* 2009), but also of changes in land-use, manipulations of bank and bed, damming, diversion and harvesting of sediments, vegetation and biota. A basic understanding of the complex linkages and interactions that are responsible for shaping and sustaining river ecosystems are essential for predicting the consequences of management decisions or interpreting data designed to monitor those consequences (Harding *et al.* 1998; Tockner *et al.* 2010).

The place and importance of macroinvertebrate communities in monitoring and understanding changes in river ecosystems has been discussed by several authors (e.g., Rosenberg and Resh 1993; Hodkinson and Jackson 2005; Haidekker and Hering 2008; Masase *et al.* 2009; Mwangi 2014). They are an important element in monitoring the ecological dynamics of lotic environments (Hynes 1970) as their composition can help ascertain the quality of freshwater ecosystems. They can act as indicators for changes in overall ecological integrity (Masase *et al.* 2009) as they are sensitive to pollution and integrate and reflect the effects of stress on the system over extended periods (Rosenberg and Resh 1993).

The functional and structural composition of macroinvertebrates varies, both spatially and temporally, in relation to local environmental factors (Tate and Heiny 1995; Nautiyal 2013), this is because families, species and genera differ from one another in their requirements and sensitivity/tolerances to environmental variability, pollution and habitat modification

(Voshell 2002; Nautiyal 2013). For instance, Ephemeroptera, Plecoptera and Trichoptera families are sensitive to disturbance, occur in clean and well-oxygenated waters (Bispo *et al.* 2006) and are considered to be indicators of good water and habitat quality (Rosenberg and Resh 1993), whereas worms, chironomids, leeches and pouch snails usually suggest a poor water quality (Abel 2002; Fisher 2003; Robertson 2006). Species with higher sensitivity require higher dissolved oxygen, while pollution tolerant species have low sensitivity and can survive with less oxygen (Fisher 2003; Robertson 2006). Most species used for biological monitoring are common and have moderate tolerance to environmental variability. Rare species that usually have narrow tolerances are often too sensitive to environmental variability, or are encountered too infrequently to reflect a general biotic response (Holt and Miller 2010).

Aquatic biomonitoring programmes have been implemented all over the world and biomonitoring programmes have been developed to survey the overall state of aquatic ecosystems in order to keep track in ecosystem change (Roux 1999). Programmes that were developed based on macroinvertebrates have been and are successfully used for the bioassessment of rivers around the world (Ollis *et al.* 2006). Macroinvertebrate communities are widely used for reasons such as being visible to the naked eye, highly diverse, well studied and easy to identify, have a rapid life cycle and their ecology and life history is relatively well understood, abundant and common with a wide distribution, live in sedentary habitats and easy to collect (Dallas and Day 1993; Chutter 1995; Rosenberg and Resh 1993).

According to Ollis *et al.* (2006), early hydrobiological studies on macroinvertebrate communities of South African major rivers laid foundations for river bioassessment in this country (e.g. Harrison 1958; Harrison and Elsworth 1958; Oliff 1960). In South Africa, the South African Scoring System version5 (SASS5) (Dickens and Graham 2002) is widely used for measuring water and habitat quality. In SASS5 quality scores are assigned to taxa based on their susceptibility or resistance to pollution or poor water quality; the lowest scores are assigned to the taxa that are resistant and the highest score to those susceptible to pollution.

The history of ecosystem structure and functioning is recognised as being important for understanding how present conditions came about (Turner 2005), how ecosystems function and defining reference conditions (Newson 2008), and is a crucial approach to ecologically-sound management (Bis *et al.* 2000; Rhemtulla and Mladenoff 2007). The spatial patterning and geographic distribution of organisms are influenced by the natural history together with ecology (Turner 1989). There are very few detailed long-term data sets in existence with which to examine large-scale temporal changes in community composition of freshwater systems (Lancaster *et al.* 1996; Ractliffe *et al.* 2007), particularly in Africa.

The Berg River is exceptional among South African rivers in that it has detailed records of macroinvertebrate community structure collected at intervals from as far back as the 1950s (Ractliffe *et al.* 2007); these data provide an invaluable record of past conditions. In the 1950s, the Berg River was considered a relatively unpolluted river (Harrison and Elsworth 1958). It also was largely unregulated, with only the impoundments being Voelvlei and

Wemmershoek dams (Harrison and Elsworth 1958). The first detailed survey of the macroinvertebrates in the system was done in 1951 by Harrison and Elsworth (1958), who visited 13 sites along the length of the river, collecting macroinvertebrate and water samples. Forty years later (in 1991), Dallas and Day (1992) repeated the Harrison and Elsworth survey, at the same sites and using a purposefully similar approach to enable comparison. There have been three subsequent surveys (Coetzer 1978; Dallas 1997; Ractliffe *et al.* 2007) of the invertebrates along the Berg River. These were augmented for this study by a survey in 2015 at the same Berg River sites following similar approaches to data collection and processing (Harrison and Elsworth 1958; Ractliffe *et al.* 2007).

This chapter presents these historical and more recent data for macroinvertebrate communities of the Berg River and evaluates the changes over time. The two hypotheses tested were:

1. Macroinvertebrate assemblages in the Berg River have changed over time, with more tolerant taxa becoming more dominant;
2. These changes are linked more closely with changes in habitat and water quality.

6.2 Study sites

Between 1958 and 2017, five surveys collected detailed data on macroinvertebrate communities along the full length of the Berg River from 13 roughly-comparable study sites, the locations of which are given in Table 6.1 and illustrated in Figure 6.1. Sub-basins were delineated along the length of the river based on the longitudinal slope (Rowntree *et al.* 2000): the mountain stream, upper foothill; lower foothill and lowland zone followed by the estuary (Chapter 3, section 3.2.1). Briefly, (with reference to Table 6.1):

Sites 1 and 2 are in the upper foothills upstream of the Berg River Dam. This area was afforested with *Pinus radiata* for many years prior to being cleared in 2005. Both sites have natural banks with indigenous trees and shrubs. The active channel was about 6 m wide and comprised riffles and shallow pools with stones (gravel, cobbles and bedrock) in and out of current. Marginal vegetation was present and there was very little aquatic vegetation.

Site 3 is in the upper foothills at Jim Fouche Bridge downstream of the Berg River Dam. This area was also afforested with *Pinus radiata* for many years and is influenced by effluents from and abstractions by the town of Franschhoek. It has modified cultivated banks covered in dense grasses and shrubs with no trees. The active channel was about 5 m wide and comprised riffles and shallow pools with cobbles and bedrock in and out of current. Marginal vegetation was present and there was no aquatic vegetation, gravel, sand nor mud.

Site 4 is in the lower foothills at Simondium, just upstream of the town of Paarl. It had unstable, highly-modified and cultivated banks with grass and very few trees. The active channel was about 10 m wide and dominated by riffles and pools with cobbles and bedrock in and out of current. Only the marginal vegetation biotope was present

Site 5 is in the lower foothills at Cecil Drift, downstream of the N1 Bridge over the Berg River. The site was stabilised and channelised banks with a mixture of indigenous and exotic shrubs and trees. The active channel was about 11 m wide and comprised riffles, pools and runs with cobbles and bedrock in and out of current. Gravels, sand and marginal vegetation were present, but there was no aquatic vegetation.

Site 6 is in the lower foothills at Daljosafat, the industrial area downstream of the Paarl Waste Water Treatment Works. It had stable and channalised banks, with a few. The active channel was about 15 m wide and comprised pools and runs where no stones in and out of current were visible. Sand, mud and marginal vegetation were present but there was no aquatic vegetation.

Site 7 is in the lower foothills at Lady Loch Bridge in Wellington. It also had stable, modified banks, but was surrounded by *Eucalyptus* spp. The active channel was about 15 m wide and comprised pools and runs where no stones in and out of current were visible. Sand, mud and marginal vegetation were present but there was no aquatic vegetation.

Site 8 is in the lower foothills at Hermon. It had unstable, modified and cultivated banks with grasses, shrubs and no trees. The active channel was about 20 m wide and comprised deep pools, runs and some riffles with stones in current. Sand, mud and marginal vegetation were present but there was no aquatic vegetation.

Site 9 is in the lowlands at Gouda. It also had unstable cultivated banks vegetated with low grasses and shrubs. The active channel was about 20 m wide and comprised deep pools, runs and some riffles with stones in current. Aquatic vegetation was absent while the stones in and out of current were not visible at this site.

Sites 10 and 11 are in the lowlands at Bridgetown and the N7 road crossing, respectively. Both sites had stable modified banks (the macro channel was cultivated) with trees and shrubs. The active channel was about 20 m wide and comprised shallow pools, runs and riffles with stones in and out of current. Gravel, sand, mud and marginal vegetation were present with little aquatic vegetation.

Site 12 is in the lowlands at Sanddrift. The banks were steep, stable and surrounded by cultivated fields with a few trees. The active channel was about 18 m wide and comprised deep pools and runs. All biotopes were absent except for a thin section of marginal vegetation.

Site 13 is in the estuary sub-basin at Kersefontein. It had stable banks but the macro channel was cultivated so there were few trees. The active channel was about 25 m wide and comprised deep pools and runs. There was marginal vegetation present but no stones, GSM and no aquatic vegetation.

The mountain stream contains the undeveloped upper reaches of the Berg River, upstream of the Berg River Dam; no sites were sampled in this zone. The upper foothill reach mostly drains minor tributaries down to the confluence with the Wemmershoek River; there are three sites within this zone. The lower foothill is located between the confluence of the Wemmershoek and Kompanjies rivers; it includes the settlements of Paarl, Dal Josafat, Wellington and Hermon. The lowland is between the Kompanjies and Sout rivers; it includes the settlements of Hermon, Gouda, Riebeek-West, Bridgetown, Mooreesburg, Hopefield and Piketberg. The estuary is the lowermost section of the basin and is influenced by tidal effects at Veldrift and Laaiplek.

Table 6.1 Location of study sites and their codes in the different studies, *bolded are dates for actual data collection

Sub-basin (see Chapter 3)	Harrison and Elsworth (1958) ***1951 to 1953**	Dallas and Day (1992) ***1991**	Dallas (1997) ***1993 and 1994**	Ractliffe *et al.* (2007) ***2003, 2004 and 2005**	This study (2015) ***2015**	Location	
						Longitude	Latitude
Upper foothills	II (1)	UBG	UBG		Site 1	19.069278	33.974537
	IIIA (3)	ABT	ABT	BRBM 1	Site 2	19.072068	33.956846
	IIIA (5)	JFB	JFB	BRBM 2	Site 3	19.034340	33.877884
Lower foothills	IIIB (9)	SMD	SMD		Site 4	18.968959	33.828032
	IIIB (10)	CDR	CDR		Site 5	18.973905	33.763512
	IIIB (11)	DJT	DJT		Site 6	18.974109	33.707663
	IIIB (12)	LLB	LLB		Site 7	18.976529	33.629653
	IV (13)	HRB	HRB	BRBM 4	Site 8	18.955967	3.4347128
Lowlands	IV 14)	GOU	GOU		Site 9	18.952915	33.256299
	IV (16)	BRD	BRD	BRBM 5	Site 10	18.860092	33.133934
	IV (18)	PKB	PKB	BRBM 6	Site 11	18.749170	32.973109
	IV (19)	SNT	SNT		Site 12	18.353600	32.535500
Estuary	V (21)	KFN	KFN		Site 13	18.200000	32.540000

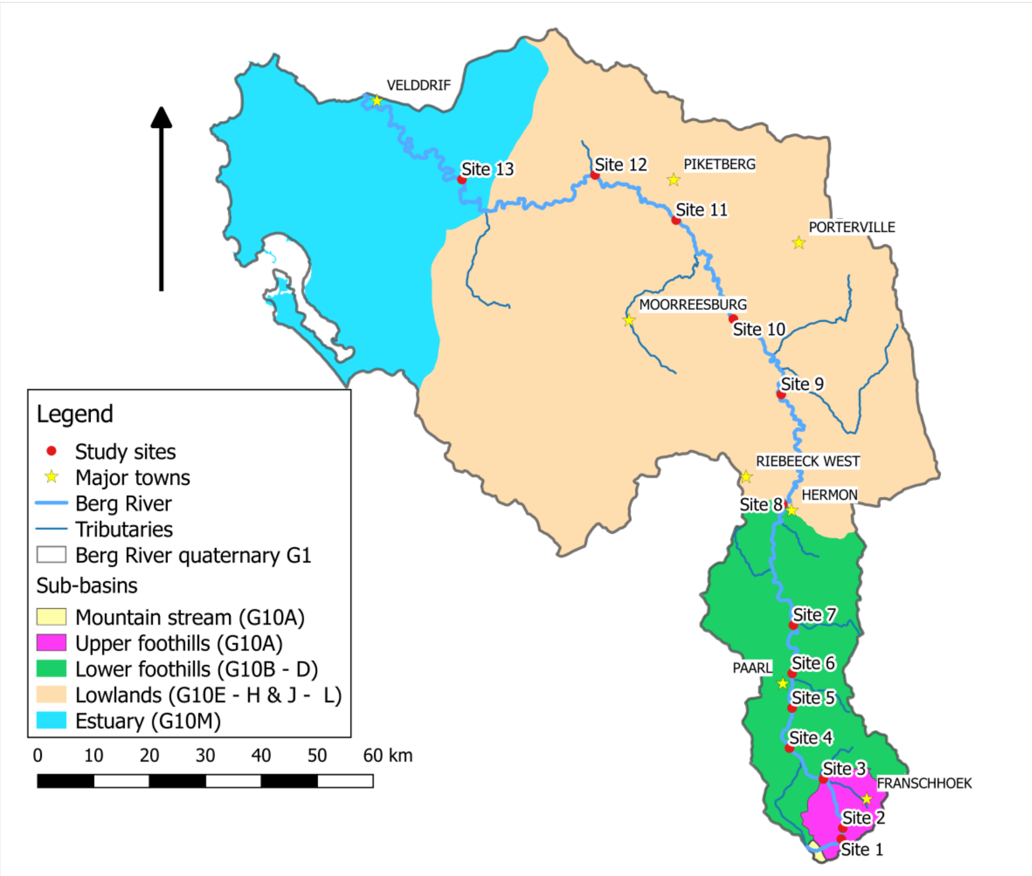

Figure 6.1 The location of the 13 study sites in the different sub-basins

6.2.1 Summary of historic land-use patterns and water-resource developments in the Berg River Basin

Historic changes in land-use in the Berg River are addressed in Chapter 3. In summary, transformation of the landscape for agriculture started as early as before the 1930's, and the extent of agricultural lands peaked at 7466.33 km^2 (83.35% of basin) in *c.* 1976-1985. Thereafter area of agricultural land declined and there was an increase in the extent of undeveloped and fallow land. Urban areas in the Berg River Basin increased ~240%, between 1951 and 2015, with concomitant increases in water use and wastewater discharge (Chapter 3; Stuckenberg 2013). The density of the urban areas also increased significantly over this time (Roy *et al.* 2006) and, in many cases, outstripped the infrastructure such as wastewater treatment works designed to service the urban settlements (e.g., Stellenbosch Municipality 2009). Over this same period, four major in-channel dams were constructed (Wemmershoek 1957; Voelvlei 1971; Misverstand 1977 and Berg River 2007) to meet water demands in and outside the basin, and the number of farm dams increased by ~290%, with the largest increases occurring in the upper (~7 in 1950s to ~125 in 2015) and lower foothills (237 to 855), and slightly less in the lowland reaches (588 to 1395). These main-channel and farms dams changed the hydrology and sediment characteristics of the river (Chapter 4), and contributed towards changes in the channel morphology and habitat availability of the river (Chapter 5).

6.3 Methods

6.3.1 Data collection methods in historic studies

Two different methods were used to collect samples in the historical studies (Box sampling and the SASS5 protocol) (Table 6.2). Harrison and Elsworth (1958) and Dallas and Day (1992) gathered macroinvertebrates from marginal vegetation using sweep nets and used square-framed benthic box samplers to gather organisms from stones in current or gravel sand and mud biotopes. At each site, two stone in current samples were collected, bottled and preserved in formalin for later laboratory identification to species level, where possible. The number of individuals of each species was recorded and the percentage composition of each species per sample was calculated. Dallas (1997) and Ractliffe *et al.* (2007) followed the SASS4 (Chutter 1994) and SASS5 (Dickens and Graham 2002) sampling protocol respectively, identifying organisms to family level only and calculating SASS specific scores according to the organisms sensitivity to poor water quality. The historic data were collected at different frequencies, either monthly or seasonally. Since data were collected during the Spring Season when flows have receded and aquatic biota has recolonised, for this study historic data from the months of Spring (September, October, November) were used.

Table 6.2 Methods used to collect macroinvertebrates in the different studies

Source	Collection method	Month of collection	Taxonomic level of identification
Harrison and Elsworth (1958)	Hand nets for marginal vegetation and square-bottomed samplers for the stony bed, both with nets of silk gauze at 23 meshes per inch.	September to November	Species
Dallas and Day (1992)	A box net sampler of 0.1 m^2 to collect animals in stones in current biotope; and sweep nets to collect animals in marginal vegetation; all with 250 µm nets.	November	Species
Dallas (1997)	SASS version 2 and 4. Hand nets of 250 mm diameter and a modified box sampler of 0.1 m^2 area.	September	Family
Ractliffe (2007)	SASS version 5. Used a normal SASS net which is a hand-held 950-µ mesh sieve attached to a 300 mm x 300 mm square frame.	September to November	Family

6.3.2 Data collection methods used in this study

Aquatic macroinvertebrates were collected in spring 2015. Two methods were used to collect samples; a box sampler and the SASS sampling protocol, so comparisons could be made to both sets of historic studies (section 6.3.1). At each site, an attempt was made to sample all habitats (SIC, SOOC, MVG, GSM) but sometimes some habitats were not present at a site (Table 6.3).

The box-samples were taken from stone in current biotopes using a 0.09m^2 square framed box sampler with 250µm mesh and collection sock. The stones in the box were first disturbed by hand, then picked up individually and brushed off using a soft toothbrush. Sampling was done to a substratum depth of *c.* 0.15 m. After all the stones had been scrubbed, the box sampler was carefully lifted and the sides of the boxes mesh and sock rinsed with water to dislodge any macroinvertebrates clinging to the net. Difficult to dislodge macroinvertebrates were picked off the net with forceps. Samples were then transferred into plastic bottles and preserved in 96% ethanol and taken to the laboratory. For sites where there was no stone in current habitat, a sample was collected from gravel sand and mud habitat (e.g., at Site 6). Box-sampling was not possible at Sites 9, 12, and 13 because the channel banks were near vertical and the channel was too deep to wade. In the laboratory, debris was picked from the sample and the organisms were sorted into families under the supervision of an aquatic macroinvertebrate expert, Ms Liesl Phigeland at the University of Cape Town. These samples were then sent to Dr Denise Schael, a macroinvertebrate taxonomist at Nelson Mandela University, where they were identified to species level. Lastly, species lists from the different sample periods were cross examined in order to eliminate duplication for species due to name changes over time.

The standard SASS5 sampling protocol (Dickens and Graham 2002) was used to collect macroinvertebrates from up to four biotopes: stone in current (SIC), stones out of current (SOOC), marginal and aquatic vegetation (MVG), and gravel sand and mud (GSM) using a 30cm square-frame hand-held net with 1mm mesh. Not all biotopes were present at all sites, for example there was no GSM at Sites 3, 4, 7, 9, 12 and 13, and no stone habitats at Sites 6, 9, 12 and 13. The Total SASS Score (SASS score), Average Score Per Taxon (ASPT) and Number of Taxa (No. taxa) were then calculated for each site. Thereafter, and contrary to the SASS procedures, the samples were transferred into plastic bottles and preserved in 96% ethanol and taken to the laboratory. Once there, they were picked/separated to family and sent for species-level identification together with the box-samples described above. This meant that the sweep samples of the marginal vegetation, when identified down to species, could be compared with the vegetation samples collected by Harrison and Elsworth (1958) and Dallas and Day (1992).

Table 6.3 Habitats that were sampled in 2015 for the two methods used (√ = collected, shaded = samples collected from sand, X = habitat absent, grey[2] = sand or mud sampled)

Sites	Stone/sand 1 (Box sampler)	Stones/sand 2 (SASS hand net)	Stones in/out current	Marginal vegetation	Gravel sand and mud
Site 1	√	√	√	√	√
Site 2	√	√	√	√	√
Site 3	√	√	√	√	X
Site 4	√	√	√	√	X
Site 5	√	√	√	√	√
Site 6	√	√	X	√	√
Site 7	√	√	√	√	X
Site 8	√	√	√	√	√
Site 9	X	X	X	√	X
Site 10	√	√	√	√	√
Site 11	√	√	√	√	√
Site 12	X	X	X	√	X
Site 13	X	X	X	√	X

For each site the Total SASS score versus ASPT were plotted along with the ecological condition bands (Table 6.5) to visually represent the scores for the macroinvertebrate assemblages over time. In SASS a quality score (between 1 and 15) that is based on susceptibility to pollution has been allocated for each taxon, a high score is attributed to sensitive organisms while lower scores are given to hard taxa (Dallas 2000). These scores were then used to evaluate how sensitivity and/or abundance may have changed over time, and between zones. In general, low Total SASS Scores reflect impairment of habitat availability, and low ASPT indicates a greater proportion of the taxa present are tolerant of poor water or habitat quality.

For the sample periods; dominant taxa were identified and their sensitivity to pollution (SASS5 quality scores) was used to define the ecological condition for sites, geomorphological zones and periods. Sensitivity weighting (Table 6.4) is a measure of

[2] If GSM was absent then samples were collected in whatever of the three was present, e.g. sand, mud or gravel.

tolerance to pollution that is given for each family in the SASS5 scoring system (Gerber and Gabriel 2002).

Table 6.4 Sensitivity weighting for aquatic macroinvertebrates as used in SASS5 scoring system

Sensitivity weighting range	Tolerance to pollution
11-15	very low tolerance
6-10	moderately tolerant
1-5	tolerant to pollution

The total SASS Score, number of taxa and ASPT were calculated for all sites. A graph of total SASS score versus ASPT was plotted, following the guidelines of Dallas (2007) in order to determine the ecological condition of the macroinvertebrate communities for each time period at all SASS sites. Upland sites included those in the source, mountain stream, transitional and upper foothill, while lowland sites included lower foothill and lowland zones. The five Ecological Categories and category boundaries for the uplands and lowlands for the South Western Cape are shown in Table 6.5. Data for years 2003, 2004 and 2005 were only collected at five of the 13 sites originally sampled.

Table 6.5 Ecological Categories for interpreting SASS data adopted from Dallas 2007a, South Western Coastal Belt

Ecological Category	Description	Boundary values for categories	
		SASS score	ASPT
A	Unmodified natural	110	6.1
B	Largely natural with few modifications	70	4.8
C	Moderately modified	53	4.4
D	Largely modified	38	3.9
E/F	Critically or extremely modified	<38	<3.9

6.3.3 Data analyses

Macroinvertebrate abundance data per sample per site were standardised (converted to percentage composition). Sites that were not sampled in at least two of the sample periods (1951/1991/2015) were removed from the analyses (Table 6.3). Since the historic data comprised two sets of data collected using different methods and identified to different levels (species versus family), different analyses were used to test whether there had been changes in the structure of the macroinvertebrate communities over time, and whether the organisms present in 2015 are more tolerant to pollution than those found in the river in earlier surveys.

Box samples: data collected in 1951 by Harrison and Elsworth (1958), in 1991 by Dallas and Day (1992) and in 2015 by this study (2015) were compared to the nearest taxon level. Data were analysed using: ANOSIM tests, CLUSTER analyses and MDS ordinations (Clarke and

Warwick 2006) to identify similarities and differences in invertebrate community structure between years [3](sample period) and sub-basins. For the ANOSIM results: R^2 values of < 0.3 meant there were no significant differences; R^2 values of 0.3 to 0.7 mean there are differences but these are not strong, and R^2 values > 0.7 indicate strong differences. The SIMPER analyses (Clarke and Warwick 2006) were used to identify taxa contributed most strongly toward differences between periods and sub-basins. For tables presenting the ANOSIM and SIMPER analysis results indicate species ; Average abundance (Av.Abund), Average similarity (Av.Sim), Similarity/standard deviation (Sim/SD), Contribution to the similarty/dissimilarity (Contrib%) and cumulative contribution (Cum.%).

SASS samples: data collected in 1993/1994 by Dallas (1997), in 2003/2004/2005 by Ractliffe *et al.* (2007) and in 2015 by this study were compared at family level. First data were analysed using presence and absence of macroinvertebrates at different periods, only sites that had data for all six periods were considered (Site 2, 3, 8, 10 and 11). ANOSIM tests were done to test for differences between periods, sites and sub-basins. CLUSTER and SIMPER analyses were also used to show typical and discriminating families. Secondly, the SASS5 protocol (Dickens and Graham 2002, Dallas 2007b) was followed, three principal indices; Total SASS score, ASPT and No. taxa were calculated for sites at each period. Sites were allocated into different Ecological Categories adopted from Dallas (2007a).

Box samples identified down to the nearest taxon for stones and sweep samples from marginal vegetation collected in 1951, 1992 and 2015 were problematic because of changes in taxonomic resolution for the different invertebrate groups over time. It is also unlikely that nearest taxon identifications will be available for routine sampling, perhaps for critical sites and important ad hoc events, but in general the nationally available database will comprise family level samples taken using the SASS protocol. For these two reasons a second round of analyses was focused on these data for this reason. SASS samples of stones in and out of current were analysed at the family level for changes over time 1992, 2003, 2004, 2005 and 2015. The 1993 samples were calibrated using the SASS5 weights and sensitivity ratings.

6.4 Results

The results first compare the box-sample data collected in 1951, 1991 and 2015 at different taxonomic levels (species, genus, family and order level). Thereafter, the SASS data collected in 1993, 1994, 2003, 2004, 2005 and 2015 are compared. Sites/periods that do not appear on the Cluster and MDS diagrams were either outliers or did not have SIC, SOOC or MVG biotope available in that year. For example, in 2015, there was no stone in current biotope at Sites 7, 12 and 13.

6.4.1 Box samples for 1951, 1991 and 2015 – nearest taxon

An ANOSIM test showed that the three periods (1951, 1991 and 2015) were different from each other (R = 0.646, P < 0.1%) and in addition the sites within periods were different from

[3] The different survey/sample years have been referred to as ‘periods’.

one another (R = 0.326, p < 0.4%). Sub-basins showed to be different from one another based on replicate sites within zones (R = 0.456, p<0.5%) while sites within the same sub-basin were not different from each other (R = -0.075, p < 90.7%). When all periods were tested for difference individually; the foothills (upper and lower) and lowland sites were significantly different from one another suggesting a downstream transition in community structure but the upper and lower foothill sites were not different from each other.

6.4.1.1 Similarity within periods

The SIMPER analysis showed that the within period similarity of the 1951 and also the 1991 samples were stronger than the 2015 samples as their average similarity percentages were higher (Table 6.6). The families Chironomidae and Simulidae both contributed strongly to the similarity of the 1951 and 1991 samples groups and at first glance appear to be less so in 2015. This was not the case however as for 2015, the genera *Polypedilum*, *Conchapelopia* and *Rheotanytarsus* are chironomids, *Simulium* is a simulid, and *Pseudocloeon* a baetid (Table 6.6). It also appears that the family Trichoptera are missing from the 2015 samples but again, even though genera from the family Trichoptera are not shown in the summary table below they were present, but their contributions toward similarity of 2015 samples was low (the sim/SD value) and not shown here. The Trichopterans of 2015 were also discriminated further down to genera and species level, whereas the 1951 and 1991 samples were not.

Table 6.6 SIMPER analysis of species contributing towards period identity

Period 1951 (Average similarity: 37.51%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Pseudocloen vinosum	1.81	8.11	1.28	21.64	21.64
Chironomidae sp.	1.50	7.51	2.11	20.01	41.65
Simulidae sp.	1.24	4.78	1.05	12.75	54.40
Baetis harrisoni	1.16	3.96	1.03	10.57	64.96
Trichoptera sp.	0.78	2.57	0.79	6.84	71.81
Baetis bellus	0.94	2.28	0.51	6.09	77.89
Period 1991 (Average similarity: 44.65%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae sp.	2.62	19.55	3.33	43.79	43.79
Simulidae sp.	1.42	7.75	1.42	17.35	61.14
Baetis harrisoni	1.11	4.18	0.76	9.37	70.50
Trichoptera sp.	0.70	3.06	0.91	6.85	77.35
Baetis bellus	0.58	2.26	0.68	5.07	82.42
Period 2015 (Average similarity: 20.18%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Baetis sp.	1.07	2.08	0.58	10.32	10.32
Polypedilum sp.	0.83	2.07	0.79	10.27	20.59
Simulium (Pomeroyellum) alcocki	0.87	1.67	0.61	8.27	28.85
Conchapelopia sp.	0.70	1.45	0.62	7.17	36.02
Rheotanytarsus sp.	0.75	1.41	0.60	6.97	42.99
Pseudocloeon sp.	0.84	1.36	0.51	6.74	49.72

Similarly, and because the table only shows the strongest contributors to group similarity, *Baetis bellus* and *B. harrisoni* were also present in 2015 but did not contribute strongly to any particular group. One other broad difference between samples from the three years was that the sim/SD values for 2015 were lower overall, relative to 1951 and 1991, and that there were no 2015 sim/SD greater than 1, which means the taxa present in 2015 across all samples were disparate and poorly represented so the relationships between samples from this year were weak.

The initial CLUSTER analysis was based on the raw sample data used above for the three periods (Table 6.6) and revealed a suspiciously unnatural separation of the 2015 samples from those collected in 1951 and 1991 (Figure 6.2). It is unlikely that there was a complete turnover of species in 20 years since 1991 even if changes in the faunal assemblages were expected in response to changes in the landscape of the river basin, flow and river channel habitats. This was because of differences in identification of taxa. These were mainly related to greater taxonomic resolution in 2015 than in the other periods, although in some cases the reverse was true. Table 6.7 gives the differences in resolution and levels at which these were lumped to ensure compatibility across the datasets. Once the data had been reduced to a common taxonomic level, the CLUSTER analysis was rerun (Figure 6.3).

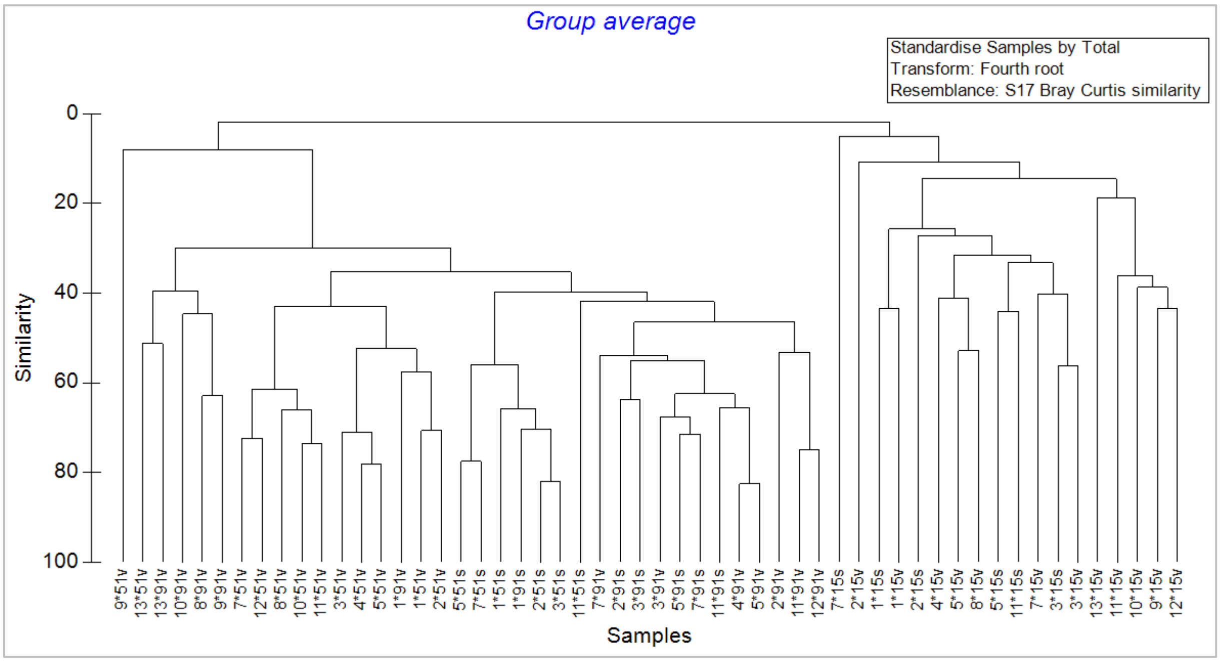

Figure 6.2 CLUSTER analysis of using raw (uncorrected) aquatic macroinvertebrate data from stone (s) and vegetation (v) habitats from sites along the Berg River collected in 1951, 1991 and 2015. Sample codes 9*51v = site 9, 1951, vegetation

Table 6.7 shows how different taxa were reduced to the nearest taxon by lumping/combining some species, genera or families in order to ensure compatibility across the three periods (1951, 1991 and 2015).

Table 6.7 Differences in taxonomic resolution and levels at which these were lumped to ensure compatibility across the periods

Order	Family	Nearest taxon 2015	1951	1991
Coleoptra	Elmidae	*Elmidae* sp.	*Elpidelmis capensis*	*Elpidelmis capensis*
			Peloriolus granulosus	*Peloriolus granulosus*
Diptera	Chironomidae	*Cardiocladius* sp.	Chironimidae	Chironimidae
		Ablabesmyia sp.		
		Chironomini sp.		
		Cladotanytarsus sp.		
		Conchapelopia sp.		
		Cricotopus kisantuensis		
		Cricotopus sp.		
		Endochironomus sp.		
		Notocladius capicola		
		Orthocladiinae sp.		
		Eukiefferiella sp.		
		Paramtriocneums sp.		
		Paratendipes sp.		
		Polypedilum sp.		
		Rheocricotopus capensis		
		Rheotanytarsus sp.		
		Stempellinella sp.		
		Tanytarsus sp.		
		Thienemanniella sp.		
		Cryptotendipes sp.		
	Simuliidae	*Paracnephia (Paracnephia)* sp.	*Simulium* sp.	*Simulium* sp.
		Simulium (Edwardsellum) damnosum s.l.		
		Simulium (Meilloniellum) adersi		
		Simulium (Metomphalus) sp.		
		Simulium (Nevermannia) nigritarse s.l.		
		Simulium (Pomeroyellum) alcocki		
Ephemeroptera	Baetidae	*Baetis* sp.		*Baetis bellus*
			Baetis capensis	
				Baetis glaucus
			Baetis harrisoni	*Baetis harrisoni*
				Baetis latus
	Tricorythidae	*Tricorythus* sp.	*Tricorythus discolor*	
Hemiptera	Corixidae	*Micronecta* sp.	*Micronecta bleekiana*	
			Micronecta diminiata	*Micronecta diminiata*

			Micronecta scutellaris	
Oligochaeta	Lumbriculidae	*Lumbriculus variegatus*	Oligochaeta sp.	Oligochaeta sp.
		Oligochaeta		
Odonata	Anisoptera	*Aeshnidae*	Anisoptera	Anisoptera
		Crenogomphus hartmanni		
		Libellulidae sp.		
		Trithemis stictica		
		Zygonyx natalensis		
	Zygoptera	*Coenagrionidae* sp.	Zygoptera	Zygoptera
		Pseudagrion hageni		
Trichoptera	Trichoptera	*Amphipsyche scottae*	Trichoptera	Trichoptera
		Athripsodes sp.		
		Cheumatopsyche afra		
		Cheumatopsyche maculate		
		Cheumatopsyche sp.		
		Cheumatopsyche thomasseti		
		Ecnomus sp.		
		Hydropsychidae sp.		
		Hydroptila cruciate		
		Leptecho sp.		
		Macrosternum capense		
		Oxyethira velocipes		
		Parecnomina resima		

An ANOSIM test from the lumped data showed that the three periods (1951, 1991 and 2015) were different from each other ($R = 0.382$, $P < 0.1\%$) however sub-basins and sites within periods were not different from one another with $R = 0.278$, $p < 3.2\%$ and $R = 0.041$, $p < 24.5\%$ respectively. When all periods were tested for differences individually; in 1951 the foothills (upper and lower) were significantly different from one another however the two sub-basins were similar in 1991 and 2015. The upper foothills were slightly different from the lowlands while strong differences were shown between the estuary and foothills suggesting a downstream transition in community structure at all periods. When sites within periods were tested individually, macroinvertebrate communities within sites were not different.

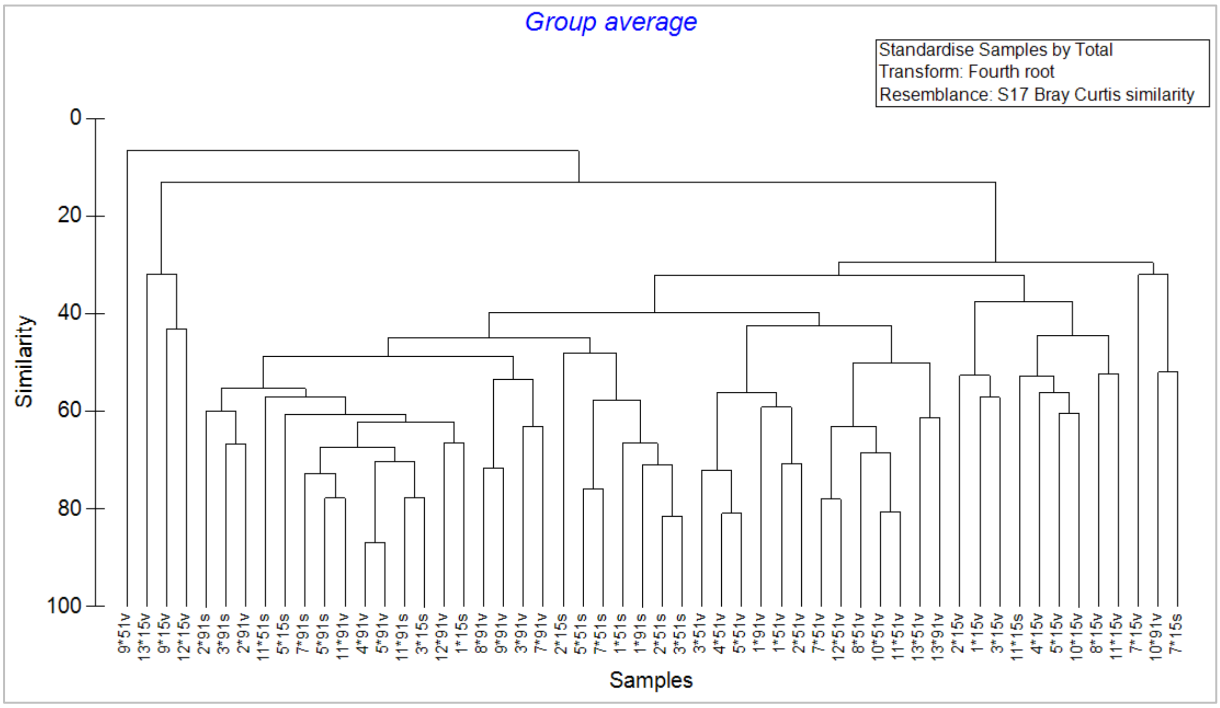

Figure 6.3 CLUSTER analysis of aquatic macroinvertebrate data from stone (s) and vegetation (v) habitats from sites along the Berg River collected in 1951, 1991 and 2015 reduced to the nearest common taxonomic level. Sample codes 9*51v = site 9, 1951, vegetation

The CLUSTER diagram derived from 1951, 1991 and 2015 data reduced to a common taxonomic level (Figure 6.3) is markedly different from that for the raw data presented in Figure 6.2. Samples collected from the 2015 MVG biotope clustered together with the upper foothill sites separating from the other sub-basins; while the SIC samples for 1991 and 2015 were grouped with samples from both SIC and MVG biotopes in other periods. For example, the 2015 vegetation samples from sites 1-8 (except for 7) clustered together with <40% similarity to the other samples (periods), whereas sites from 1951 and 1991 had clustered together. This suggests that the communities at those sites changed significantly between 1991 and 2015. The 2015 stones in current samples on the other hand, clustered with other sites from the historic periods at different sub-basins. For instance the 2015 Site 1, 3 and 5 stone in current samples were more than 60% similar to the 1991 SIC and MVG samples from lower foothill and lowland sites. The 2015 Site 2 SIC clustered with 1951 SIC sites 1-3, 5 and 7 with just below 50% similarity.

Table 6.8 SIMPER analysis of species contributing toward period identity- lumped data

Period 1951 (Average similarity: 43.33%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Baetis sp.	1.91	10.1	1.93	23.31	23.31
Pseudocloen vinosum	1.81	8.78	1.3	20.25	43.56
Chironomidae	1.5	8.2	2.1	18.93	62.49
Simulium sp.	1.24	5.16	1.06	11.91	74.4

Trichoptera	0.78	2.8	0.79	6.46	80.86
Nais sp.	0.62	1.77	0.6	4.09	84.95
Period 1991 (Average similarity: 51.55%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae	2.62	20.87	3.47	40.48	40.48
Baetis sp.	1.78	11.97	2.5	23.21	63.69
Simulium sp.	1.42	8.35	1.41	16.2	79.89
Trichoptera	0.7	3.27	0.91	6.35	86.24
Nais sp.	0.52	1.55	0.47	3	89.24
Zygoptera	0.37	0.82	0.39	1.58	90.82
Period 2015 (Average similarity: 33.07%)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae	1.89	10.73	1.56	32.44	32.44
Trichoptera	1.46	5.84	1.08	17.66	50.1
Simulium sp.	1.08	3.57	0.76	10.8	60.9
Baetis sp.	1.09	3.15	0.58	9.53	70.43
Pseudocloeon sp.	0.84	1.62	0.51	4.91	75.34
Zygoptera	0.74	1.53	0.51	4.62	79.96
Mesovelia vittigera	0.61	1.11	0.46	3.37	83.33

A SIMPER analysis showed that communities of invertebrates were different in each period ($R = 0.334$ $p < 0.1\%$). The within year average similarity of samples within periods were strongest for samples collected in 1991 (51.55%) and 1951 (43.33%) but weakest in 2015 (33.07%) (Table 6.8). Macroinvertebrate communities from the Baetidae, Chironomidae, Simulidae and Trichoptera families were dominant during all periods. The species *Baetis* sp. and *Pseudocloen vinosum* (all Baetidae) were most dominant in 1951, while Chironomidae and *Baetis* sp. were dominant during the 1991 period. 2015 was dominated by Chironomidae and Trichoptera (Table 6.8). With baetids populating the 1951 sample while the 1991 and 2017 communities were dominated by chironomids; this indicates a decrease in water quality since chironomids are more common in polluted water. Other groups also pointed to declines in habitat and water quality, for instance, plecopterans, which are sensitive to pollution and prefer clean stony beds, were only represented in 1951; caenids, which prefer slower flowing areas often associated with a sandy bottomed channel, and mesovelids, which live in slow-flowing areas with marginal vegetation, were only represented in 2015.

There were no differences between the macroinvertebrates collected at the upper and lower foothill sites, but there were differences between those collected from the upper foothill and the lowland sites ($R = 0.333$, $p < 0.57$). A Simper analysis showed that these differences were largely due to the exclusive presence of the ephemerellids *Lithogloea harrisoni* and *Lestagella pennicillata* and the leptophlebids *Aprionyx peterseni* and *Castanophlebia calida,* a helodid beetle and the stone fly *Aphanicerca* spp., in the upper foothills, all of which are more sensitive to pollution than the other six organisms found only in the lower foothills; the bugs *Micronecta* sp., *Apassus* sp. and *Gerris swakopnesis,* the mayflies *Caenis* sp. and *Pseudocloeon* sp., and the snail *Ferrissia,* all of which are more tolerant to pollution.

6.4.1.2 Dissimilarity between periods

The SIMPER analysis showed that samples collected in 2015 were highly dissimilar from both 1951 and 1991 samples; with dissimilarity percentages higher than 66% (Table 6.9). Taxa driving change between periods appear to be Baetids, Chironomids, and Simulids.
Samples collected in 1951 were 59.97% different from those of 1991, with *Pseudocloen vinosum*, Chironomidae and *Simulium* sp*.* contributing highly to the difference. Higher dissimilarities were seen between the 1951 and 2015 samples at 74.37%, with *Pseudocloen vinosum, Baetis* sp*.* and Trichoptera contributing strongly to the difference. The overall dissimilarity between the 1991 and 2015 samples was 66.75%, with *Baetis* sp*.*, Trichoptera and *Simulium* sp*.* contributing strongly to the difference.

Table 6.9 SIMPER dissimilarity analysis of taxa contributing toward period separation

Periods 1951 and 1991 (Average dissimilarity: 59.97%)					
	Av.Abund 1951	Av.Abund 1991	Av.Diss	Diss/SD	Contrib%
Pseudocloen vinosum	1.81	0.25	6.6	1.6	11.01
Chironomidae	1.5	2.62	4.52	1.49	7.54
Simulium sp.	1.24	1.42	3.49	1.24	5.83
Baetis sp.	1.91	1.78	3.28	1.29	5.47
Nais sp.	0.62	0.52	2.69	1.04	4.49
Micronecta sp.	0.29	0.49	2.58	0.68	4.31
Trichoptera	0.78	0.7	2.58	1.29	4.3
Periods 1951 and 2015 (Average dissimilarity: 74.37%)					
	Av.Abund 1951	Av.Abund 2015	Av.Diss	Diss/SD	Contrib%
Pseudocloen vinosum	1.81	0	6.28	1.65	8.44
Baetis sp.	1.91	1.09	4.4	1.36	5.92
Trichoptera	0.78	1.46	3.64	1.41	4.9
Simulium sp.	1.24	1.08	3.34	1.23	4.5
Chironomidae	1.5	1.89	2.8	1.07	3.76
Pseudocloeon sp.	0	0.84	2.54	0.9	3.41
Zygoptera	0.29	0.74	2.47	1.02	3.32
Nais sp.	0.62	0	2.21	0.83	2.98
Periods 1991 and 2015 (Average dissimilarity: 66.75%)					
	Av.Abund 1991	Av.Abund 2015	Av.Diss	Diss/SD	Contrib%
Baetis sp.	1.78	1.09	4.62	1.39	6.92
Trichoptera	0.7	1.46	4.05	1.47	6.06
Simulium sp.	1.42	1.08	3.72	1.16	5.57
Chironomidae	2.62	1.89	2.92	1.44	4.37
Pseudocloeon sp.	0	0.84	2.74	0.9	4.11
Zygoptera	0.37	0.74	2.72	1.02	4.08
Micronecta sp.	0.49	0.33	2.41	0.65	3.62
Mesovelia vittigera	0	0.61	2.03	0.81	3.04
Nais sp.	0.52	0	1.92	0.84	2.87

6.4.1.3 *Summary of main findings – nearest taxon analyses (lumped data)*

The 1951 and 1991 samples were still strongly similar to one another than to those of 2015; however there were reduced differences between 2015 community structure and that of these historic periods when compared to those shown by the first SIMPER using the unconsolidated data (Table 6.6). The similarities between the 1951 and 1991 samples were given by taxa of *Baetis* sp., and Chironomidae while Chironomidae and *Simulium* sp. had strongly contributed to the similarity of the 1991 and 2015 periods. *Pseudocloen vinosum* was highly dominant in 1951, contributing strongly to the dissimilarity of 1951 to both 1991 and 2015 community. Trichoptera was highly dominant in 2015, contributed highly to the difference dissimilarity of 2015 to both 1951 and 1991 communities. ANOSIM suggested that macroinvertebrate communities of the foothills had changed after the 1951 sample, i.e there were no differences between taxa collected from the upper and lower foothill sites during 1951.

6.4.2 SASS family level data

For this analysis only sites that had data for all six years were used (Sites 2, 3, 8, 10 and 11). A one-way ANOSIM test showed there were no differences between the 1993 and 2004, 1994 and 2003 and between 2003, 2004, 2005 and 2015 in any respective combination (Global $R = 0.215$; $p < 0.2\%$). There were differences between 1993 and 1994 ($R = 0.428$; $p<3.2\%$); 1993 and 2003 ($R = 0.364$; $p < 0.8\%$), 2005 ($R = 0.514$; $p < 0.8\%$) and 2015 ($R = 0.368$; $p < 1.6\%$); and 1994 and 2004 ($R = 0.356$; $p < 4.8\%$), 2005 ($R = 0.613$; $p < 0.8\%$) and 2015 ($R = 0.609$; $P < 1.6\%$).

A one-way ANOSIM for differences between sub-basins showed that the upper foothill and lower foothill were different from one another ($R = 0.409$; $p < 0.3\%$) and the upper foothill was different from the lowlands ($R = 0.267$; $p < 0.3\%$), but the lower foothill was not different to the lowlands ($R = 0.101$; $p < 21.5\%$).

A one-way ANOSIM for differences between sites showed there were no differences between replicate sites within sub-basins; for instance Site 2 and 3 in the upper foothill were not different from one another ($R = 0.16$; $p < 8.2\%$), Site 8 in the lower foothill was not different from the two lowland Sites 10 and 11 ($R = 01$; $p < 46.3\%$; and $R = 0.041$; $p < 31\%$ respectively).

The CLUSTER analysis showed Sites 2 and 3 in the upper foothill for all years separating from the rest of the sites, with Sites 8 and 11 of 1993 and 1994 being outliers in this group. Sites 8, 10 and 11 of the 2000s (2003, 2004, 2005) and 2015 data also clustered together with more than 40% similarity.

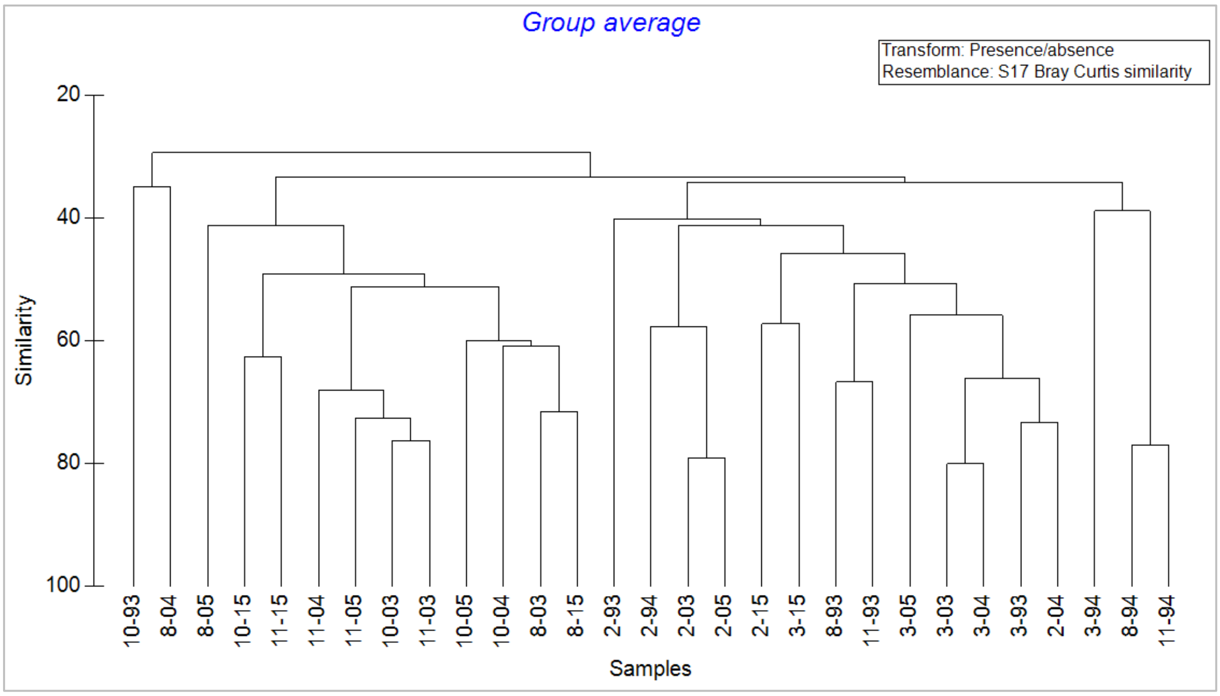

Figure 6.4 CLUSTER analysis of aquatic macroinvertebrate samples collected from stone and vegetation habitats in 1993, 1994, 2003, 2004, 2005 and 2015

6.4.2.1 Similarity within periods

Group similarity of presence and absence of macroinvertebrates families at sites between periods was comparable over time, between 40 and 50% each year (Table 6.10). Only families with Sim/SD values greater than 1 are shown as these are the strongest contributors to group/period similarity. Chironomids were strong contributors every year and Simulids were also in all years but not 2004 and 2015. Oligochaetes and Coenagrionids were present in the 2000s and 2015 as strong contributors but not in the 1990s data set. Naucorids were only strong contributors in 2015 while Caenids, Corixids, Velids and Gyrinids were stronger in the 2000s.

Table 6.10 SIMPER similarity analysis of taxa contributing toward group identity

Period 1993 (Average similarity: 46.55)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Baetidae 1sp.	1	8	4.74	17.18	17.18
Chironomidae	1	8	4.74	17.18	34.37
Aeshnidae	0.80	4.41	1.12	9.48	43.85
Corixidae*	0.80	4.41	1.12	9.48	53.32
Simuliidae	0.80	4.41	1.12	9.48	62.8
Period 1994 (Average similarity: 41.99)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae	1	11.36	4.06	27.06	27.06
Simuliidae	1	11.36	4.06	27.06	54.11
Period 2003 (Average similarity: 47.94)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae	1	4.96	8.48	10.35	10.35
Simuliidae	1	4.96	8.48	10.35	20.70

Caenidae	0.80	3	1.14	6.26	26.96
Coenagrionidae	0.80	3	1.14	6.26	33.23
Corixidae*	0.80	3	1.14	6.26	39.49
Veliidae/Meso-veliidae*	0.80	3	1.14	6.26	45.76
Gyrinidae*	0.80	3	1.14	6.26	52.02
Leptoceridae	0.80	2.73	1.16	5.70	57.72
Elmidae/Dryopidae*	0.80	2.73	1.16	5.70	63.43
Period *2004* (Average similarity: 36.16)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Corixidae*	1	5.75	6.14	15.91	15.91
Oligochaeta	0.80	3.62	1.13	10.01	25.92
Hydropsychidae 1sp	0.80	3.32	1.13	9.18	35.1
Veliidae/Meso-veliidae*	0.80	3.26	1.13	9.03	44.13
Chironomidae	0.80	3.26	1.13	9.03	53.15
Period *2005* (Average similarity: 43.33)					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Gerridae*	1	6.13	7.08	14.15	14.15
Simuliidae	1	6.13	7.08	14.15	28.3
Gyrinidae*	0.80	3.97	1.15	9.17	37.47
Coenagrionidae	0.80	3.53	1.14	8.15	45.62
Chironomidae	0.80	3.41	1.15	7.88	53.5
Period *2015 (*Average similarity: 42.44*)*					
	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Oligochaeta	1	6.77	23.42	15.95	15.95
Chironomidae	1	6.77	23.42	15.95	31.89
Coenagrionidae	0.80	4.14	1.16	9.76	41.65
Naucoridae*	0.80	4.01	1.16	9.44	51.09

6.4.2.2 Dissimilarity between periods

1993 was different to the other years in that there were Aeshnids and Potamonautids that were not recorded 1994, 2005 and 2015, which along with Libellulids (1994) and Gomphids (2005) drove differences between these years (Table 6.11). Other main differences post the 1990s, were the presence of Lymnaeids and Oligochaets, which are usually associated with slow flows, fine sediments and detritus. From 2003 to 2005 caseless Hydrophyschids were dominant; they are typical of fast flowing rivers under stones. In 2003 and 2015, and an increase in the abundance and dominance of beetles and bugs was noted, they are normally associated with aquatic and marginal vegetation in pools and/or backwaters and flow flowing rivers, over this time, including, Hydraenids, Gerrids, Dyticids, Gyrinids, Velids, Notonectids and Naucorids.

Table 6.11 SIMPER dissimilarity analysis of taxa contributing toward period separation

	Period	Present in					
		1993	1994	2003	2004	2005	2015
Compared to	1993		Baetidae >2sp	Ancylidae Lymnaeidae* Oligochaeta Hydropsychidae 2sp Hydroptilidae Baetidae >2sp Hydraenidae*		Gerridae* Lymnaeidae* Dytiscidae* Baetidae 2sp	Naucoridae* Baetidae 2sp Notonectidae* Hydropsychidae 2sp
	1994	Baetidae 1sp Aeshnidae Potamonautidae* Libellulidae			Oligochaeta Veliidae Dytiscidae*	Gerridae* Gyrinidae* Libellulidae Dytiscidae* Veliidae Lymnaeidae* Oligochaeta	Oligochaeta Naucoridae* Gyrinidae* Notonectidae* Veliidae Hydropsychidae 2sp
	2003	Baetidae 1sp					
	2004						
	2005	Gomphidae Aeshnidae Potamonautidae*					
	2015	Aeshnidae Potamonautidae*					

6.4.2.3 Summary of main findings - family level

The main differences between years was driven by Baetids, Aeshnids, Gomphids and Libellulids, along with Potamonautides in 1993 and 1994, when compared to 2003, 2004 and 2005 where there were more bugs and beetles being characteristic in the samples collected, along with Oligochaets, a situation that did not change much into 2015. Although the relative abundance of the different groups in the later years changed over time, the organisms present were from similar guilds, being suited to slower flowing water, marginal and aquatic vegetation, and comprising carnivores and detritus feeders. Differences in sub-basins were a little different from those given by the analysis of samples collected using the box sampler whereby upper and lower foothill were the same but both upper/lower foothills were different not same as the lowlands.

6.4.3 SASS5 Score and ASPT

For each sample period; dominant taxa were identified and their sensitivity to pollution was used to define the ecological condition for sites, sub-basin and periods. Sensitivity weighting (Table 6.4) is a measure of tolerance to pollution that is given for each family in the SASS5 scoring system (Gerber and Gabriel 2002). The SASS and ASPT scores for each site and each period are shown in Table 6.12, and plotted against each other for ecological category demarcations following Dallas (2007a). In general, low Total SASS scores reflect low habitat diversity, and low ASPT scores indicate poor water or habitat quality.

All three metrics studied (SASS score, number of taxa and ASPT), differed among biotopes, with highest scores consistently recorded in the stones biotope, while lowest SASS scores and fewest taxa were recorded in the gravel sand and mud biotope. On average SASS scores were highest in the 2000s (2003/2004/2005) and lowest in 2015 (Table 6.12). The 2015 SASS scores and the ASPT at Sites 1-3 are lower than those in 1993 and in 1994, while the scores at the more downstream sites are similar for all three times. From the total SASS score and ASPT for sampled sites (Table 6.12), there was no obvious/gradual longitudinal trend visible from these data; however, the scores downstream of Daljosafat are generally lower than upstream of this point. Sites in the upper foothill (Sites 1-3) had higher SASS scores and ASPT for all periods while Site 6 (Paarl) and Site 9 (Gouda) had the lowest. Across all periods the SASS scores, number of taxa and ASPT were always higher at Sites 10 and 11 when compared to other neighbouring sites at the lower foothill and lowlands (Table 6.12). When comparing sites that had the full six years repeat data (Sites 2, 3, 8, 10 and 11), it was clear that higher diversity was recorded in 2003 with more than 20 taxa at each site with exception to Site 8 at Hermon (Figure 6.5).

Table 6.12 Average Score per Taxon (ASPT) and Total SASS score at each site and each sampling period

Sites	Total SASS score						Total number of taxa						Total ASPT					
	1993	1994	2003	2004	2005	2015	1993	1994	2003	2004	2005	2015	1993	1994	2003	2004	2005	2015
Site 1	144	111				35	16	13				7	9	8.53				5
Site 2	77	133	163	146	151	84	9	13	21	21	22	14	8.55	10.23	7.70	6.90	6.80	6
Site 3	131	62	170	165	79	71	20	9	25	25	14	14	6.55	6.88	6.80	6.60	5.60	5
Site 4						55						12						4.5
Site 5	36	56				59	8	10				13	4.5	5.60				4.50
Site 6	22	21				27	5	7				7	4.4	3				3.80
Site 7	59	64				30	13	15				6	4.53	4.26				5
Site 8	76	32	65	69	54	67	15	7	14	14	12	16	5.06	4.57	4.60	4.90	4.50	4.10
Site 9	30	33				30	9	9				7	3.33	3.66				4.20
Site 10	42		110	58	75	73	9		22	14	16	17	4.66		5	4	4.60	4.20
Site 11	67	37	104	66	107	50	13	8	20	15	19	14	5.15	4.62	5.20	4.40	5.60	3.50
Site 12	47					28	10					8	4.70					3.50
Site 13	5					3	2					1	2.50					3

Shifts between Ecological Categories within sites are presented in Table 6.13, presenting only the five sites that had data from all sample periods. Total SASS score and ASPT were plotted against each other in order to classify sites into different Ecological Categories A to E/F (after Dallas 2007a). Sites located in the upper foothills remained unmodified (Category A) during the historic periods but in 2015 the condition changed to largely natural with few modifications (Category B) at all periods. Sites in the lower foothills and lowlands were generally classified as either largely natural with few modifications (Category B) or moderately modified (Category C) between periods. In 2015 sites were generally classified into lower Ecological Categories when compared to the previous years; Site 11 in particular moved from a Category B in 2005 to a Category D (largely modified). In 2003 all sites were in a better ecological conditions (Categories A and B). Change is evident over time. For instance, at Site 8 at Hermon, the condition of the macroinvertebrates in 1993 was largely that of natural state (Category B) and by 1994 the site condition had dropped to a moderately modified (Category C). In 2003 the site condition improved to an unmodified natural (Category A) but then deteriorated to a Category B in 2004 which thereafter dropped to a Category C in both 2005 and 2015.

Table 6.13 Ecological Category/ condition of study sites over time (after Dallas 2007a)

Site	Sample period					
	1993	1994	2003	2004	2005	2015
Site 2	A	A	A	A	A	B
Site 3	A	A	A	A	B	B
Site 8	B	C	A	B	C	C
Site 10	C	-	B	C	B	B
Site 11	B	C	B	C	B	D

According to the 2015 data none of the site ecological conditions had improved; sites had either remained within the same Ecological Category (Site 3, 8 and 10) or deteriorated (Site 2 and 11).

The number of taxa was different at all sites and periods with the 1993/94 period having a lower diversity in overall. The highest diversity was recorded for samples that were taken between the 2003 – 2005 sampling periods. Across all three sampling periods, Site 8 at Hermon particularly had the lowest number of taxa (Figure 6.5).

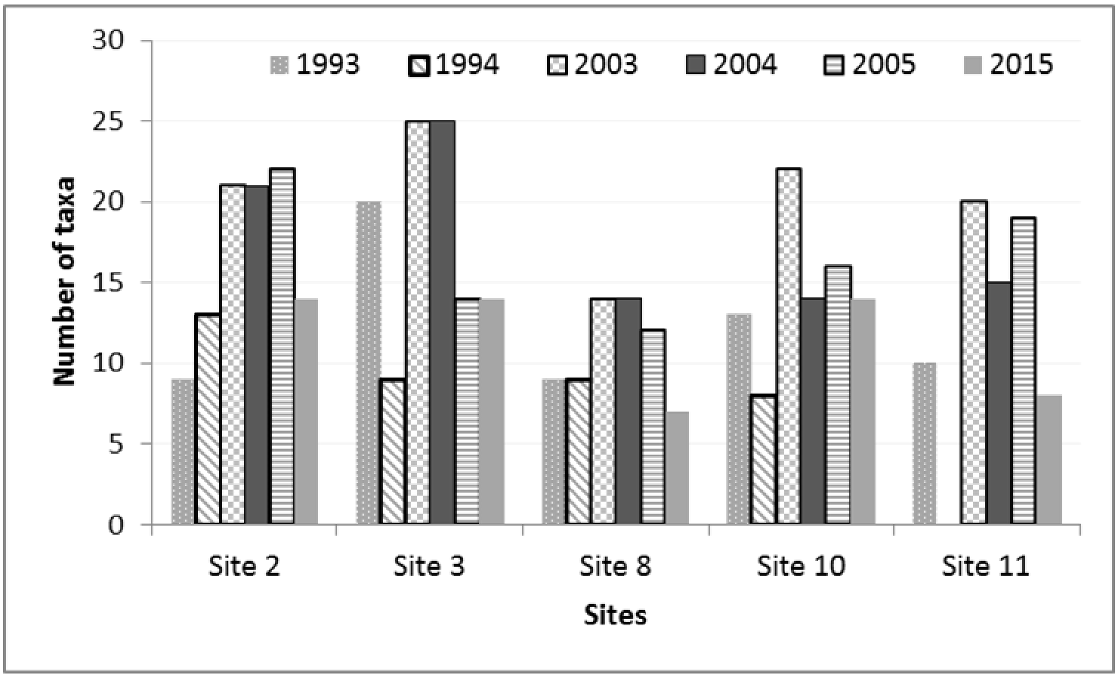

Figure 6.5 **Change in the number of taxa over time for sites that had data for all six years**

6.5 Discussion

In the 1950s, the Berg River was considered relatively unpolluted and unregulated (Harrison and Elsworth 1958), with the only major impoundments being Voelvlei and Wemmershoek dams (Harrison and Elsworth 1958), neither of which was situated on the mainstem. There is little doubt that this has changed over time.

The initial results of the various historical macroinvertebrate surveys analysed in this chapter suggested that there have been colossal changes in the macroinvertebrate community structure of the Berg River between the mid-1900s and now. The separation of 2015 so strongly from the other periods was partly an artefact of different levels of resolution in taxonomic identifications between the periods, which resulted in Type 1 differences in the resultant analysis. For instance, Chironomidae and Simulidae contributed strongly to the similarity of the 1951 and 1991 datasets, and alothough they occurred in the 2015 samples, they were recorded as *Polypedilum*, *Conchapelopia, Rheotanytarsus,* and *Simulium,* and so were not recognised as Chironomidae and Simulidae. There were also differences in how Trichoptera and especially cased-caddises were discriminated taxonomically; for example, the families Leptoceridae, Petrothrinicidae and Sericostomatidae were reported as one in 1951 and 1991 but as three families in 2015. Similarly, Odonata, Zygopterans and Anisopterans were separated into more taxonomic groups in 2015 than in 1951 and 1991. Not all greater resolution was in the 2015 identifications, for instance, *Beatis harrisoni* in 1951 and 1991 were lumped under *Beatis* sp. in 2015. These sorts of differences are understandable; the animals in the box samples collected in 1951 were identified by Arthur Harrison at a time when there were few taxonomic guides available; the bulk of the 1991 samples were identified by Dr Helen Barber at the Albany Museum Grahamstown, 40 years later, and the 2015 samples were identified by Dr Denise Schael at NMU, nearly 70 years after Harrison and Elsworth collected and processed their samples. However, the initial analysis of raw data underline the extreme importance of interrogating historic data for taxa where identifications may have changed, either lumped or splitting or through name changes. The need for this sort of interrogation also considerably increases the time and expertise needed to assess historical biotic data, and certainly seems to suggest that the 'noise' created by species level data (e.g., Bournaud *et al.* 1996) can arise for numerous reasons.

The corrected data yielded less stark, but still significant, differences between the periods. The differences were most marked between 1991 and 2015, but were also evident between 1951 and 1991. It is tempting to assign the residual differences to variations between taxonomists, differences in sampling techniques and/or a difference in the ability to discriminate different taxonomic groups. Although known changes in the taxonomy of South African invertebrates were accounted for, to the extent possible, in the correction of the data, and the researchers involved in the collection of samples were either experienced macroinvertebrate ecologists and/or were accompanied by experienced macroinvertebrate ecologists, it is almost certain that the 70% difference between 1951/1991 and 2015 macroinvertebrate communities for the box samples at least partially still reflects differences in species identifications and/or sampling techniques. However, two main factors suggest that the differences between the periods when box data were collected may not be solely a result of variation in taxonomic identification. The first is that these differences extend as far as family level, and the second is that the SASS scores and ASPT (collected using different methods (SASS4 and SASS5) and at different times) support the notion that the health of the ecosystem has declined significantly over the last 20 years (1993-2015).

Identification of macroinvertebrates to the taxonomic level of Family is a relatively straightforward proposition, and misidentification is unlikely provided the sampler has had some experience with identifying macroinvertebrates (Nahmani *et al.* 2007; Jones 2008; Austen 2016). For instance, the more recent samples were all collected by accredited SASS practitioners, who must pass an exam on family-level identification to receive such accreditation. It is thus unlikely that the families were misidentified. Genus-level identifications are, of course, a lot more difficult but not for experienced specialist taxonomists, which all of the people involved were. Thus, while the question highlights the importance of the researchers being experienced and accredited, it is unlikely that much of the difference shown between the sample periods using the corrected data is attributable to differences in taxonomic identification and/or sampling techniques.

In SASS, scores are assigned to families based on their susceptibility or resistance to poor quality water or habitat. These scores were assigned on thorough consultation with experienced South African macroinvertebrate ecologists and based on presence/absence scores of data collected at different sites with different water and habitat qualities (Chutter 1994, 1995, 1998; Dickens and Graham 2002). It is thus fair to assume that the SASS scores and ASPTs[4] reflect the prevailing conditions in the river at, and/or immediately prior to, the time of sampling with some degree of assurance (Dewson *et al.* 2007; Garcia-Roger 2011). The lowest scores are assigned to the taxa that are more tolerant to water or habitat degradation and the highest scores to those that only occur where water and habitat qualities are good or excellent (Dickens and Graham 2002).

The results from the various sampling periods show that Sites 1 and 2 in the upper foothills were dominated by sensitive taxa in 1993/4 and 2003, which indicates that the water and habitat quality was good and very well oxygenated (Bredenhand 2005), but these were either much reduced or absent in 2015 and the SASS scores and ASPTs were correspondingly lower. The reason for this change could be that the catchment upstream of both sites was deforested in 2006 (EWISA 2007) as part of the site preparations for the Berg River Dam, constructed in c. 2007. Site 3, also in the upper foothills, was dominated by sensitive taxa in 1993/4, but in 2005 and 2015, these were either much reduced or absent. This was probably because the Franschhoek Wastewater Treatment Works were working at ~300% beyond capacity and discharging effluent that did not meet the DWS discharge water quality standards (www.dwa.gov.za) directly into the Berg River over this time (Stellenbosch Municipality 2009), which may have masked the activities upstream related to the construction of the Berg River Dam. The SASS scores and ASTPs for Sites 5 and 6[5] were similar and low for all the sampling periods. The SASS results for the lower sites were somewhat less clear, but in general the samples from these sites were dominated by Chironomidae, Hydropsychidae, Simulium, Baetidae and Oligochaeta, which are fairy tolerant of poor water and habitat conditions (e.g., Quinn *et al.* 1997). For instance, *Chironomus* spp. are known to occur in greater abundance in areas with environmental stress as the genus is able to colonise water with low oxygen concentration (Johnson *et al.* 1993). The increased numbers of these taxa can be attributed to organic pollution as a result of enrichment and sedimentation caused by agricultural activities in the riparian areas (Dallas 2005; Bredenhand 2005) but are also to be expected downstream as the river becomes wider, flows more slowly and deposition of sediment increases (Resh *et al.* 1995). Thus, despite some mid-period improvements in SASS and ASPT scores (e.g. 2003 - Sites 3, 8, 10 and 11), the SASS data clearly indicate a decline in overall river condition from 1993 to 2015.

It is thus a distinct and defensible possibility that the changes reported by the data are true, and that macroinvertebrate communities in the Berg River have changed significantly at a species, genus and family level over the past 70 years. Furthermore, in order for that change to have happened, there must have been a substantial change in either water quality or habitat quality, or both. Furthermore, the river ecosystem at Sites 1, 2 and 3, is considerably less ‘healthy’ in 2015 than it was 22 years ago (in 1993). Differences in community composition at sites were also reported by Coetzer (1978) who collected macroinvertebrate samples of the Berg River in 1974. A significant decrease in the number of ephemeroptrans particularly baetids, was reported; Coetzer (1978) added to this *Baetis*

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[4] The number of biotopes sampled is positively correlated to the SASS score and number of taxa of the site, while ASPT has a negative relationship (Dallas 2007b).

[5] Site 4 was not sampled in 1993/4 or 2005.

glaucus and *Pseudocloeon maculosum* were reportedly missing at all sites by 1974. All sites showed a higher percentage of chironomids when compared to the 1951 samples (Harrison and Elsworth 1958).

For the box sample results, the levels of separation between sampling periods differed depending on the taxonomic level of identification that was applied. The species differences (~98%) shown between 1951/1991 and 2015, respectively, were extensive, with no overlaps in the species present today in the river (2015) when compared to those in the past. For the 1951 analyses, Dallas and Day (1992) reported a clear separation of biotope groups while the distinction between vegetation and rocky substratum was less clear in 1991. A more unclear separation of biotopes was also seen for the 2015 samples. Percentage composition for species in Dallas and Day (1992) were lower than those reported by Harrison and Elsworth (1958). Only 2% of the species that were collected in the previous surveys was present in 2015. Mayflies *Baetis harrisoni, Lithoglea harrisoni, Castanophlebia calida* and *Aprionyx peterseni* together with *Lymnae collumela* snails were the only species that were collected across all years; with very low abundances of these present in 2015.

Similarly high dissimilarity percentages (up to 66%) were obtained when samples collected in 1951 were compared to those of 1991. As the analyses moved up the taxonomic groups from species, to genus, to family and then order, the within-year sample similarity increased and correspondingly the dissimilarity between sample periods decreased. This is related to successively less discriminant taxon groups providing less scope for separating group-based differences from one another.

Importantly, the *family-level* box data showed significant differences between 1991 and 2015 for all the sites, whereas the SASS scores, which also rely on family-level taxonomic identification, for 1993 and 2015 were not extremely different, particularly those from the lower reaches (Table 6.12). The reasons for this are that the box samples recorded more families than did SASS (e.g. Site 8: 1991/3 data yielded 14 families for box and 12 families for SASS, with seven common between the two; 2015 data were 14 families for box and 15 families for SASS, with nine common)[6], plus the analysis of the box sample data included consideration of the abundance of each family, which has been shown to be a strong distinguishing factor in comparisons of community composition. Given that misidentification of families is less likely that misidentification of species or genera, and the similarity in the number families in each, it is reasonable to assume that the box sample results are reflecting changes in habitat and water quality missed by the SASS samples.

There are two main lessons here for the collection and use of macroinvertebrate data in long-term monitoring programmes. The first is that species level data, which is time-consuming and expensive to collect, may be susceptible to changes in species recognition parameters over time. We are not sure of the extent to these are documented, if at all, but even if they are well documented and readily available, it means that any historical datasets are likely to require 'updating' to new species names, which is difficult if not impossible

[6] This analysis is somewhat complicated by the fact that the 1991 box and 1993 SASS data were collected at different times, although they were collected by the same researcher (Dr Helen Dallas).

without access to the original samples. This means that if such analyses are to be done at species level, the financial and training support for taxonomy and sample curation will need to be considerably higher than at present. Even in a perfect world, discontinuity between teams undertaking this work is likely to lead to researcher-driven differences in communities that are difficult, if not impossible, to separate from degradation-driven differences in community structure. The second is that it is imperative that the SASS scoring be augmented with some consideration of abundance, or at least dominance, of the various families. More recent SASS data collections include a coarse estimate of abundance, but this is not used in the scoring, nor in most cases is it reported outside of the raw data (which are rarely available to follow-up researchers); although the REMP database does make provision for the inclusion of these data. The observation that the sensitivity (and usefulness) of SASS scores could be considerably enhanced through the inclusion of some consideration of abundance is not new and was also noted by Brown (1997). It was not possible to test this in here as abundance data were not collected in the historic SASS sampling.

7 A proposed framework for the use of historic data to support the River Eco-Status Monitoring Programme

Note: This chapter is a joint effort and was co-authored by Rozwi Magoba, Karl Reinecke, Alison Joubert and Cate Brown. Rozwi Magoba was responsible for the bulk of the work in Section 7.3, and contributed meaningfully to all the other sections.

7.1 Introduction

Water is an indispensable natural water resource that is fundamental to the quality of life and environment (DWA 2004). In South Africa, the government is the custodian of this scarce resource and the National Water Act 36 of 1998 (NWA 1998) is the principal legal instrument relating to water resources. Two other important pieces of legislation also govern river basin activities in South Africa: the National Environmental Management Act (NEMA; Act 107 of 1998) and the Conservation of Agricultural Resources Act (CARA; Act 43 of 1983). Each of these laws is aimed at protecting the integrity of the country's rivers so that they can continue to support the livelihoods of South Africans for generations to come. The departments that oversee this legislation share the responsibility of managing and monitoring river condition, and each monitors river ecosystems in some way. The focus of this chapter, however, is the NWA and thus the monitoring activities of the Chief Directorate: (CD: Water Ecosystems) at the Department of Water and Sanitation (DWS).

The NWA makes provision for the 'Reserve', which is defined as: 1) the quantity and quality of water required to satisfy Basic Human Needs (BHN) by securing a basic water supply as prescribed under the Water Services Act 1997 (Act no. 108 of 1997) for people who rely on water from the relevant water resources, and; 2) the quantity, quality and distribution in time of water to protect aquatic ecosystems so as to ensure ecologically-sustainable development and use of the relevant water resources. The latter is referred to as the "Ecological Reserve".

The Ecological Reserve stipulates the pattern and volume of a river's flow regime in order to facilitate its maintenance in some prescribed Resource Quality (NWA; Act 36 of 1998). Resource Quality is defined as the quality or all aspects of the water resource, including (NWA; Act 36 of 1998):

- the quality, pattern, timing, water level and assurance of instream flow;
- the water quality, including the physical, chemical and biological characteristics of the water;
- the character and condition of the instream and riparian habitat; and
- the characteristics, condition and distribution of the aquatic biota.

The NWA also makes provision for the Classification (NWA; Act 36 of 1998) of water resources based on their ecological condition and the requirements of other users, and the setting of Resource Quality Objectives (RQOs; NWA; Act 36 of 1998), which encompass target conditions for physical, chemical and biological characteristics of rivers, wetlands and estuaries, and the Ecological Reserve needed to meet them. Once set, the RQOs must be monitored to determine whether or not they are being met.

Since the early 1990s, the DWS has funded and managed studies throughout South Africa to provide the information necessary for Classification and setting of the RQOs for the country's rivers. The basic scientific information is now available for most major rivers in the country (www.dwa.gov.za) and Classification has either been completed or is underway in six out of nine Water Management Areas (WMAs). During this same period the River Eco-Status Monitoring Programme (REMP)[7] was developed and implemented nationally. The REMP focusses on sites established during Reserve determination studies, which collate and synthesise ecological and hydrological information at each site, and provide preliminary benchmarks for ongoing monitoring.

Within the constraints inherent in rolling out a national programme with limited budgets and skilled personnel, the data that underlie the Ecological Reserve determinations and Classification, combined with those that are generated by the REMP, provide a useful basis against which to evaluate both the implementation of the Reserve and the efficacy of the underlying flow recommendations. However, as the focus of the DWS shifts from Reserve determination and Classification to full-scale implementation of the Ecological Reserve, it is expected that there will be increased pressure to defend the Ecological Reserve and to show that it is working; particularly since the Ecological Reserve has direct bearing on the water available for abstraction for other uses. This is likely to highlight the inherent difficulties of predicting and monitoring the relationships between flow and ecosystem condition, not least because flow is not the only variable responsible to dictating ecosystem condition (Davies *et al.* 2015). It stands to reason, therefore, that interpretation of monitored changes in rivers and their relationship to flow should be contextualised within an understanding of a full a suite as possible of potential impacts on the river ecosystem. This is the central concept of Integrated Water-Resource Management (DWAF 2004), which "*promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems*" (GWP 2000).

The history of an ecosystem's structure and functioning is vital for understanding how present conditions came about (Turner 2005), how the ecosystem functions, and for defining its reference condition (Newson 2008). As such, collating and analysing the past is crucial to ecologically-sound management (Bis *et al*. 2000; Rhemtulla and Mladenoff 2007). The distribution of organisms at large and small scales is influenced by their natural and anthropogenic histories (Turner 1989). Historical data gathered from different sources such as pollen cores, tree rings, old land survey records, written accounts of early travellers, cadastral maps, aerial photographs and oral interviews (Egan and Howell 2001) have all

[7] Previously called the River Health Programme (RHP).

been used to uncover ecosystem drivers (Rhemtulla and Mladenoff 2007). Today, these are augmented by the high resolution, freely available images on Google Earth® over large geographic regions that provide current and historical views of land-use, and offer an inexpensive and highly accessible means of assessing character, composition and patterns in rivers (Johnson and Host 2010). Increasingly, they are also augmented by the analysis of other high spectral satellite images such as SPOT and Landsat TM (e.g., Klemas 2014).

This information can be used to identify and document past and current pressures on the system, establish the historical context for the aquatic ecosystems, enhance the understanding of how these responded to past pressures, and; ensure that monitoring data (e.g. REMP data) are interpreted within an understanding of past pressures on the system.

With this in mind the objectives of this chapter are to:

- Link the environmental and biological information presented in the previous chapters through a meta-analysis aimed at identifying which of the environmental variables are the main drivers of changes seen in the biotic communities over time
- Provide a conceptual framework for the use of historic data to support the River Eco-Status Monitoring Programme

7.2 The River Eco-Status Monitoring Programme

The objective of the original RHP was to report on the condition of rivers in South Africa, but this evolved over time, and the renamed REMP is expected to also answer the question, "*Is the Ecological Reserve being implemented correctly and achieving the desired results*?".

The methods currently used in REMP focus on calculating scores for six indices through a general assessment of cause and effect:

- Geomorphological Assessment Index (GAI, Rowntree *et al.* 2013);
- Physico-chemical Assessment Index (PAI, Scherman 2008);
- Fish Response Assessment Index (FRAI, Kleynhans 2008);
- Macroinvertebrate Response and Assessment Index (MIRAI, Thirion *et al.* 2008);
- Vegetation Response and Assessment Index (VEGRAI, Kleynhans *et al.* 2007);
- Index of Habitat Integrity (Kleynhans *et al.* 2008).

The Ecoclassification manuals (Kleynhans and Louw 2007) describe the tasks undertaken and provide MS Excel-based rule models to calculate the scores for these indices relative to hypothetical reference conditions from A to F, where A represents close to natural and F is a critically modified condition. By and large, the methods are qualitative and subjective, and rank how flow and non-flow related impacts are expected to change conditions from natural. None of the methods explicitly state the period that defines "natural conditions" or the scale at which impacts are to be assessed. However, based on experience, different practitioners collect data in different ways to derive and calculate flow-linked relationships that are then used to score and rank the identified impacts that change the outcomes of the models.

The methods were derived using data and flow-linked relationships from past Reserve studies the assessments but do not take account of specific Reserve requirements at a site when scoring conditions. Rather, the focus is whether ecological conditions have changed from natural and/or from previous sampling times, and whether the reasons for change are perceived as being flow-related or not.

The GAI is an assessment of changes in connectivity between the river and hillslopes of the surrounding basin; sediment supply and features of the river channel and floodplain; and channel stability. There is no internal database of natural conditions expected so a combination of maps, aerial images and data collected in the field are used to construct a hypothetical reference condition and to calculate the GAI score using:

- a qualitative assessment of the flow conditions at the time of a site visit;
- consideration of changes in MAR and flood frequency (if available);
- site photographs, plan view sketches and surveyed channel cross-sections;
- changes in channel width and depth;
- a description of the dominant bed material in sediment size classes;
- a qualitative assessment of channel and bank morphology (habitat) such as presence of benches, floodplain and terraces, and steps, cascades, pools, riffles, rapids, backwaters, bars, secondary channels and islands.

The PAI is an assessment of how measured water quality parameters, collected during the site visit or downloaded from the DWS water quality database (http://www.dwa.gov.za/iwqs/wms/data/000key.asp) differ from the concentrations recommended for domestic and agricultural use programmed into the PAI database. The main variables used to calculate the PAI score are:

- inorganic salts (sodium Na, calcium Ca, magnesium Mg, chlorinee Cl, sulphate SO_4);
- nutrients (phosphate PO_4 and Total Inorganic Nitrogen TIN);
- dissolved oxygen;
- pH;
- turbidity;
- temperature;
- toxic substances.

The FRAI is an assessment of changes in habitat conditions for fish based on a qualitative assessment of flow velocity and depth during a site visit and how this differs to that expected in an un-regulated, naturally shaped and vegetated river channel. It also compares the frequency of occurrence of exotic and indigenous fish species collected from the site to an internal database of those expected to calculate the FRAI score using:

- the extent of flow-depth classes for a river reach (slow-deep, slow-shallow, fast-dep and fast-shallow);
- the frequency of occurrence of species recorded versus those expected to occur.

The MIRAI is an assessment of changes in habitat and water quality conditions that affect invertebrates. It also compares the frequency of occurrence of invertebrate families to an internal database and calculates the MIRAI score using:

- a qualitative assessment of river dimensions, sediment type and habitat types present;
- measured basic water quality parameters, such as pH, conductivity and oxygen concentration;
- the frequency of occurrence of macroinvertebrate families present versus those expected to occur.

The VEGRAI is an assessment of changes in the cover and species composition of the riparian vegetation. There is no internal database of indigenous or exotic species expected to occur so all and any records available are used to construct a hypothetical reference condition and to calculate the VEGRAI score using:

- site photographs, a plan view sketch and a surveyed channel cross-section;
- a species list of plants present and their location in the channel or on the bank;
- an account of the extent of exotic plant species present.

The IHI is an overall assessment of river condition based on the scores calculated for each component above and for different river types according to:

- whether the river is perennial or non-perennial and whether the site is situated high up or lower down the rivers longitudinal profile, using either the NFEPA or national SANBI GIS covers database;
- a qualitative measure of whether the channel width and depth has changed from natural, based on a site visit or the GAI assessment;
- a qualitative assessment of natural, degraded, cultivated and urban land-use from notes made during a site visit or viewed in Google Earth®, and;
- an assessment of changes in base flow and floods from an analysis of the flow record if possible, or qualitatively based on gut feel.

The REMP data are written up in unpublished DWS reports and are incorporated into the published 'State of the Rivers Reports', both of which summarise the ecological condition scores per component for rivers across the country. The REMP data can also be accessed for use in other studies by downloading it from the REMP website (http://www.dwa.gov.za/iwqs/rhp/naehmp.aspx).

7.3 Meta-analysis of historic data

7.3.1 Potential contribution of historical data to REMP assessments

The environmental and biotic data generated in Chapters 3 - 6 provide important insights into the sorts of historical data and information that are available for a basin; which of those can be readily accessed, and the effort required to process different kinds of data.

One of the main, and most obvious, advantages of historical data are that they assist with quantifying baseline conditions that exclude a significant portion of 'modern' impacts because they often extend back to before these occurred. This is important because many of the Reserve and REMP assessments are based on deviation from some reference condition, usually natural (Kleynhans and Louw 2007). For instance, PAI is based on the difference between measured water quality and water quality parameters recommended for domestic and agricultural use. Although useful in many situations, these surrogate parameters will fail in others, such as in the Doring River in the Western Cape, which is naturally saline from spring until first winter flush (DWA 2011). An analysis of the historical water quality would allow replacement of these broad surrogates with values that better reflect the geological and hydrological characteristics of the Doring Basin. Many of the other REMP indices require comparison against lists of species that are '*expected to occur*' for which basin-specific calibration can only be achieved with some understanding of the natural history of the basin.

7.3.1.1 Land-use

Reserve projects typically consider land-use broadly as an inference for impacts at the Reserve sites and to calculate a habitat integrity score. This involves ranking the importance of different types of land-use at the site.

Mapping land-use over time (Chapter 3):

- quantified the type and extent of changes and when these took place in the basin;
- provided insight into the consequences of these changes, for example:
    - a switch from dryland farming to irrigated crops requires increased water supply that necessitates increased abstraction from rivers and storage in farms dams;
    - the expansion of urban areas leads to an increase in the release of treated and untreated sewage effluent into rivers;
    - gentrification converts working farms to lifestyle estates that require water for landscaped gardens, golf courses and recreational dams;
- provided a context to understand changes in the 'naturalised' flow record that predate the observed flows.

The mapping exercise was time consuming as it involved working with old analogue maps but the historic assessment of land-use only needs to be done once per basin and the level of effort should be viewed in that light, as updates of change going forward using electronic and georeferenced maps will be quicker. The broad changes in land-use across the basin over time were useful to understand the river's flow regime and also to inform possible cause and effect for changes in river channel shape, habitats and aquatic fauna down the line.

7.3.1.2 Hydrology and rainfall

Ecological Reserve determinations model 'present day' and naturalised flow using modelled hydrological data[8] and/or rainfall to describe relationships between the flow regime and the river ecosystem. These modelled data are not useful for determining whether or not the Reserve is being met and/or historical trends in hydrology, which require the measured data. The patching and analysis of measured hydrological records for the Berg River was time consuming and technically difficult but provided invaluable insight into the history of changes in flow prior to and after the Reserve was set 20 years ago. Once the historical flow regime had been patched and the methods for doing so established, updating the flow regime going forward and the use of these data to retrospectively check whether the Reserve maintenance flows are being met is fundamental to the entire monitoring programme.

In summary, patching the historic daily discharge:

- meant that the observed daily flow record could be analysed and compared to observed rainfall records to discriminate whether changes in flow were taking place in response to a dry period or whether they were due to abstractions, which:
    - provided temporal changes in flow on an hourly, seasonal and annual basis that informed when changes in flow took place and how long these persisted;
    - facilitated the analysis of ecologically relevant flow indicators for the length, duration and magnitude of flow in the wet and dry season and the occurrence and number of intra- and inter-annual floods;
- contextualised the relative wetness or dryness of sample years;
- allowed for comparisons between actual hydrological records with the hydrological requirements for the Ecological Reserve.

7.3.1.3 Water quality

The Berg River Monitoring Programme included a comprehensive assessment of historical water quality records for the Berg River and so these were not reassessed in this study (Ractliffe *et al.* 2007). Collation and assessment of historic WQ data holds similar benefits to those outlines for the assessment of historical hydrological data.

7.3.1.4 Channel morphology and riparian vegetation

Reserve projects record and describe a number of channel features at each site using a combination of surveyed cross-sections, channel maps, aerial images and site photographs. Sites are selected taking cognisance of the reach in which they are located and how this relates to the diversity of reaches basin wide using a desktop approach but the data collected are site specific. This information is used to calculate a reference and present day scores for the geomorphological condition and how this is likely to change in response to flow.

Mapping historical features of the river channel: described changes at a reach scale providing insight to how parts of the river respond differently from one another and to the

[8] Based on measured data.

tributaries. Changes were quantified numerically and visually over time providing evidence for changes only previously described, such as the change from a braided channel to one with a floodplain below the Berg River Dam, the loss of multiple river channels at the Twenty-fours River and the reduction in sand bars downstream of Misverstand Dam.

Reserve projects describe the plant species and communities present taking cognisance of the position where each species is located in the channel and on the banks. This site specific information is used to calculate a reference and present day scores for the condition of the riparian vegetation.

Mapping historical features of the riparian vegetation:

- described changes at a reach scale, providing insight to how parts of the river have changed over time in response to natural and human-induced impacts;
- quantified these changes numerically and visually over time providing evidence for changes only previously described qualitatively, such as:
    - changes in riparian area and floodplains along the tributaries;
    - changes from woody riparian zone in 1938 to a riparian zone comprised of grasses and shrubs following removal of the forests below the Berg River Dam by 2003;
    - the invasion of the riparian area by woody alien trees at Hermon in 2009 and subsequent clearing of the trees by 2017.

The collation and analysis of channel changes were relatively straightforward, did not require GIS programs or skills, and was well worth the effort to understand past river conditions. Since many changes predate the hydrological records it is worth sourcing the oldest aerial images available and including these in the analysis, but later analysis using Google Earth® was much simpler. The time-line of changes can also be updated from time to time, using future Google Earth® images.

7.3.1.5 Aquatic fauna

Reserve projects record and describe the presence and abundance of aquatic habitats available to invertebrates and fish at each site using a combination of site photographs and qualified statements. This site specific information is used to calculate a reference and present day score for the condition of the aquatic fauna and how this is likely to change in response to flow. No data on fish were interrogated in this project and the comments made below pertain to aquatic macroinvertebrates only since work on other aquatic fauna, such as fish, reptiles, birds and mammals, are also worked through at species level.

The comparison of historical records of invertebrate species revealed:

- Data adjustments were needed in order to account for differences in the methods of collection, the methods of analysis and taxonomic names changes.
- Little gain for the effort and expense incurred to work at a species level mainly because the life history information is difficult to obtain or not available, which limits

interpretation. However this also means that this effort is an investment to future research.

- Family-level data were easy to compare with other data since most were collected using the SASS5 method and there are much more life history data available to create flow-linked guilds to aid interpretation.

The historic dataset of invertebrates from the Berg River provided an invaluable opportunity to investigate changes at a species level over time and are unlikely to be available at most other rivers. Family level data, however, are routinely collected through the REMP and almost every freshwater ecological study conducted nationally. These data were collated into the Rivers database (http://www.dwa.gov.za/iwqs/rhp/naehmp.aspx) up to 2015. There is currently a project underway to incorporate the data from the old BioBase with the Rivers database, which will take the historical family level data back to the early 1990s (Dallas *et al.* 2011) and will be available to inform monitoring efforts in catchments around the country in a newly established Freshwater Biodiversity Information System housed at the Freshwater Research Centre (www.frcsa.org.za).

7.3.2 Data collation

The data compiled for Chapters 3 - 6 are spread across different scales of space and time, and therefore needed to be re-arranged for use in a meta-analysis. To do this, the data were divided according to three sub-basins (upper foothills, lower foothills and lowlands) and three periods (roughly >50 years ago, 20-30 years ago and 0-10 years ago). With respect to the three sub-basins: land-use was reported for each sub-basin, one gauging weir, one channel change site and one invertebrate site was selected to represent each sub-basin (Figure 7.1 and Table 7.1).

Table 7.1 Data sets used in the meta-analysis

<table>
<tr><th colspan="2" rowspan="2">Chapter</th><th rowspan="2">Variables tested</th><th colspan="3">Samples used</th></tr>
<tr><th>Upper foothills</th><th>Lower foothills</th><th>Lowlands</th></tr>
<tr><td rowspan="4">Environmental data</td><td>Land-use (Chapter 3)</td><td>Area of dryland farming, orchards and vineyards, plantation, total agricultural area, undeveloped land and built up areas (Table 3.3).</td><td>1955/1965
1996/2005
2006/2015</td><td>1955/1965
1996/2005
2006/2015</td><td>1955/1965
1996/2005
2006/2015</td></tr>
<tr><td rowspan="2">Hydrology (Chapter 4)</td><td rowspan="2">MAR, Do, Dd, Dq, Fo, Fq, Fv, Fd, Ddv, T1dv, Fdv, C1w, C2w, C3w, C4w and C5 (Table 4.6).</td><td>G1H004-77 (1959-2016)</td><td>G1H036 (1966-2016)</td><td>G1H013 (1964-2013)</td></tr>
<tr><td>IFR Site1</td><td>IFR Site 3</td><td>IFR Site 4</td></tr>
<tr><td>Channel and riparian change (Chapter 5)</td><td>Degree of braiding and sinuosity. Area of sandbanks and bars, floodplain, and channel and riparian zone. Proportion of woody to non-woody vegetation. Continuity of riparian vegetation (Section 5.2.3).</td><td>Berg River at Franschhoek River tributary</td><td>Berg River at Hermon</td><td>Berg River at Twenty-fours River tributary</td></tr>
</table>

Biological data	Macroinvertebrates (Chapter 6)	Taxa in marginal vegetation	Site 3 (Franschhoek)	Site 8 (Hermon)	Site 10 (Bridgetown)

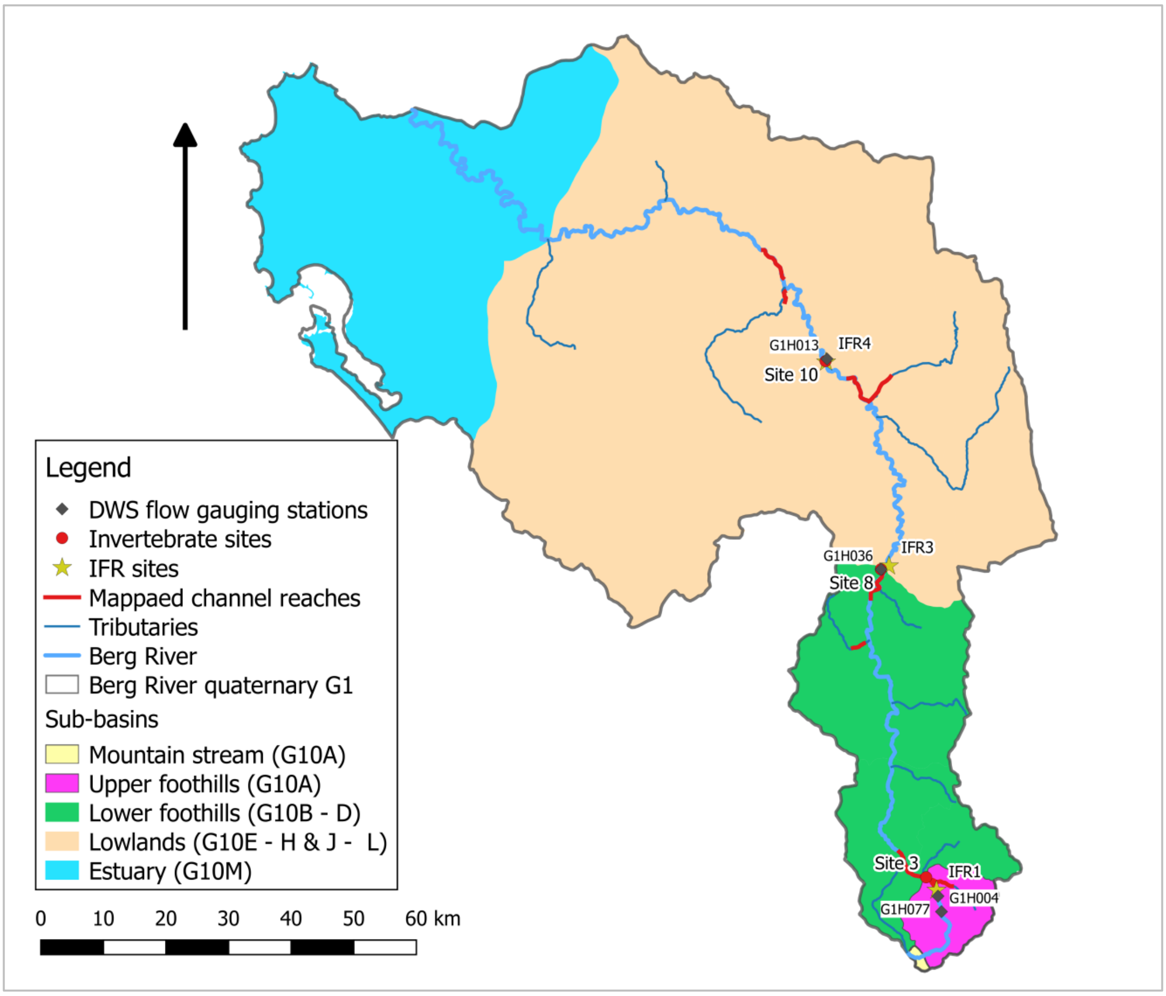

Figure 7.1 Map of the study area. Instream Flow Requirement (IFR) sites

With respect to the periods of assessment, samples were allocated to the nearest of the three periods as shown in Table 7.2. Thus, for land-use the 1955/1965 data were allocated to the >50 year period, the 1996-2005 were allocated to the 20-30 year period, and the 2006-2015 data were allocated to 0-10 year period. For hydrology flow data collected in 1959-1965 was averaged and allocated for the >50 year period, data from the 1996-2005 samples were allocated to the 20-30 year period and data from the 2006-2015 samples were allocated to the 0-10 year period. For channel change, the data generated from 1938 images were allocated to the >50 year period, data from the 2003/06/09 images were allocated to the 20-30 year period and data from the 2017 images were allocated to the 0-10 year period. For invertebrate data, data from the 1951 samples (Chapter 6) were allocated to the >50 year period, data from the 1991 samples were allocated to the 20-30 year period and data from the 2015 samples were allocated to the 0-10 year period.

Table 7.2 Time periods of samples allocated to the three time periods in the meta-analysis

Data	> 50 years	20-30 years	0-10 years
Land-use samples	1955-1965	1996-2005	2006-2015
Hydrology samples	1959-1965	1996-2005	2006-2015
Channel change samples	1938	2003/06/09	2017
Macroinvertebrates samples	1951	1991	2015

7.3.3 Data analysis

Multivariate statistics in PRIMER (Clarke and Warwick 2006) were used to assess the relationship between the environmental (land-use, hydrology and channel morphology) and the biological variables (community composition of aquatic macroinvertebrates). Bray-Curtis similarity coefficients were calculated separately for the environmental and biological data sets and the results plotted using a Multidimensional Scaling Ordination (MDS).

The BEST procedure in PRIMER was used to find the best match between the macroinvertebrate sample patterns and that of the environmental variables associated with those samples. There are two sub-analyses in BEST (Clarke and Gorley 2006): BIOENV searches for a full suite of possible correlations between the environmental and biological data while BVSTEP is a stepwise search for an optimal set (Clarke and Gorley 2006).

For the environmental data, the Euclidean distance of normalised data was used for the resemblance matrix, while the Bray Curtis similarity was applied for the biological data in PRIMER (Clarke and Gorley 2006). A BEST correlation analysis was run using a full suite of environmental data (land-use, hydrology and channel structure change), after which the effects of the hydrology and the combined land-use and channel change were tested independently. For the BEST correlation results: Rho values of <0.3 meant there were no significant differences; Rho values of 0.3 to 0.6 mean there are differences but these are not strong, and Rho values >0.7 indicate strong differences. MDS of both environmental and biological data were used to show similarities of variables between sub-basins and periods.

7.3.4 Results

The environmental data separated out into sub-basins with a low stress value of 0.05, (Figure 7.2), with the strongest cohesion shown in the upper foothill group, which means that location was the driving variable.

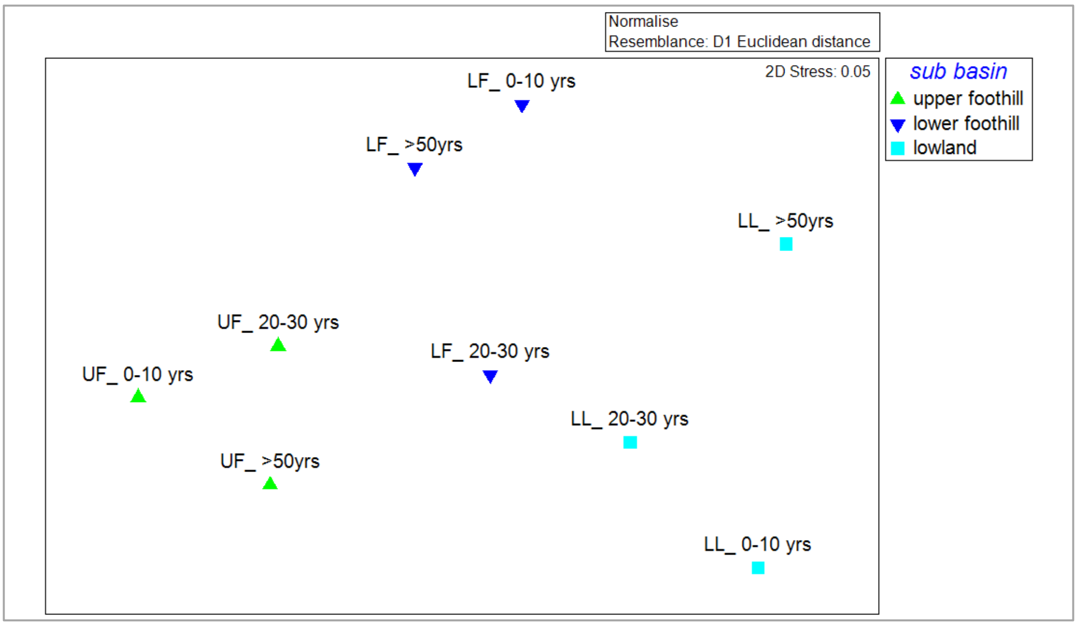

Figure 7.2 MDS showing distribution of the environmental data

The biological data separated out into time periods at a low stress value of 0.08, which means that time was the driving variable (Figure 7.3).

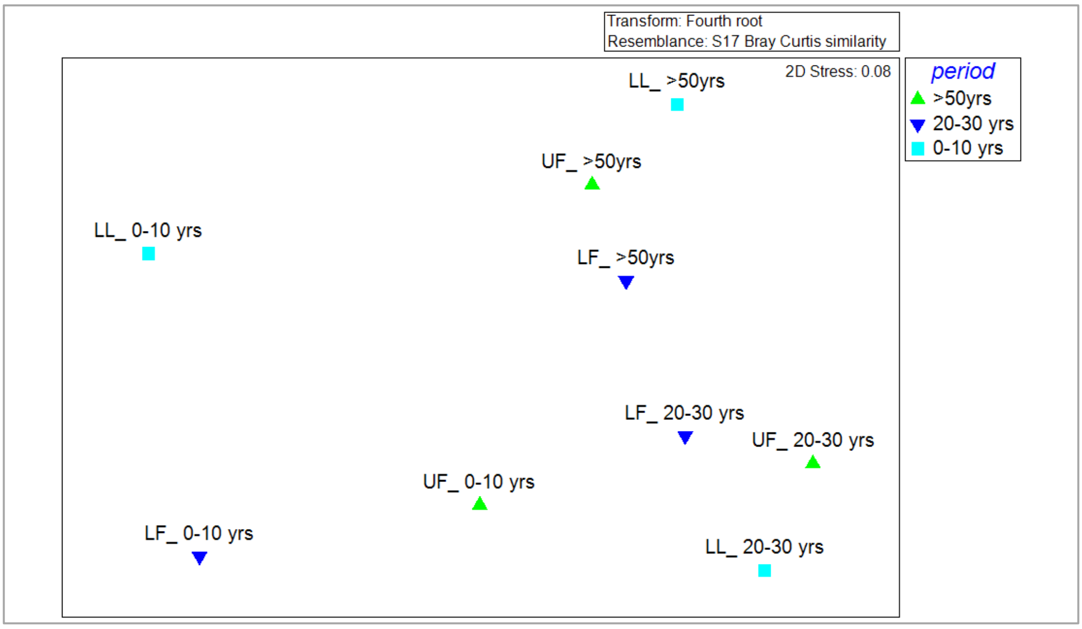

Figure 7.3 MDS showing distribution of the biological data

When the environmental variables were tested against the biological variables using BIOENV (Table 7.3), the area of plantations, area of undeveloped land, the extent of braiding, maximum 5-day average discharge in the wet season (m^3/s) and the daily average volume in the dry season (Mm^3/d) were most strongly related to changes in macroinvertebrates (Rho =

0.696). The BVSTEP analysis identified area of undeveloped land and extent of braiding as most strongly related to changes in macroinvertebrates (Rho = 0.671).

When the land-use and channel morphology variables were run without the hydrology, the results were much the same. The Rho values were high (> 0.671 for both BIOENV and BVSTEP), and area of plantations, area of undeveloped land and area of channel and riparian zone, and extent of braiding were identified as the main drivers of macroinvertebrate community structure.

When the hydrological indicators were tested alone, the BIOENV and BVSTEP returned low values (Rho ~ 0.38), which indicate that the hydrological variables on their own were less strongly related to changes in the aquatic macroinvertebrates than changes in land-use and the channel morphology. Nonetheless, the BVSTEP indicated that main hydrological drivers were the minimum 5-day average discharge in the dry season (m^3/s), the maximum 5-day average discharge in the wet season (m^3/s) and the daily average volume in the dry season (Mm^3/d). This was echoed in the BIOENV results that also identified Mean Annual Runoff (MAR), duration of the dry season, the daily average volume in the dry season, and the number of Class 1 intra-annual floods in the wet season.

Table 7.3 Results of the BEST analysis

Variables tested	Rho value	Driver identified
Land-use, hydrology and channel morphology combined		
BIOENV	0.696	Plantations, undeveloped land, maximum 5-day average discharge in the wet season (m^3/s), daily volume in the dry season (Mm^3/d), the proportion of river channel that is braided
BVSTEP	0.671	Undeveloped land, proportion of river channel that is braided
Land-use and channel morphology combined		
BIOENV	0.662	Plantations, undeveloped land, proportion of river channel that is braided
BIOENV	0.636	Undeveloped land, extent of the channel and riparian area, the proportion of river channel that is braided
BVSTEP	0.671	Undeveloped land, the proportion of river channel that is braided
Hydrology		
BIOENV	0.381	Mean Annual Runoff (m^3/s), duration of the dry season (days), minimum 5-day average discharge in the dry season (m^3/s), daily average volume in the dry season (Mm^3/d), the number of class 1 intra-annual floods in the wet season
BIOENV	0.379	Mean Annual Runoff (m^3/s), minimum 5-day average discharge in the dry season (m^3/s), maximum 5-day average discharge in the wet season (m^3/s), the number of class 1 intra-annual floods in the wet season
BVSTEP	0.378	Minimum 5-day average discharge in the dry season (m^3/s), maximum 5-day average discharge in the wet season (m^3/s), daily average volume in the dry season (Mm^3/d)

7.4 Framework for contextualizing REMP outputs

Biological patterns, such as those in habitat availability or quality and/or macroinvertebrate communities, are generated by processes acting over various temporal and spatial scales. Thus, meaningful interpretations of REMP should be underpinned by consideration of the influence of factors at a wider spatial scale and over a longer period than is possible in routine REMP monitoring (e.g. Thompson *et al*. 2001). Such information and data are used to support interpretation of site specific biological monitoring and to help establish cause-and-effect relationships.

The requisite to include consideration of these factors is not new, and many (but not all) reports and presentations using REMP data have taken due consideration of their spatial and temporal context. The purpose of the proposed framework is to encourage more widespread and systematic incorporation of large scale long-term data into monitoring and interpreting implementation of the Ecological Reserve; and to provide method statements for collating and analysing historical data for land-use, rainfall, hydrology, channel change and historic faunal surveys.

The spatial and temporal spread of historical data for land-use, rainfall, hydrology, channel change and riparian vegetation, climate change, and historic faunal surveys, relative to data collected as part of REMP field surveys, are illustrated in Figure 7.4. The potential use of these data to inform the calculation of the indices required for REMP, and/or their analysis and interpretation is illustrated in Figure 7.5, based on the assessment of value in Section 7.3.1.

Collecting and collating historical data can be time consuming but the resultant information is an invaluable contribution to understanding the basin and interpreting monitoring outcomes, and the process only needs to be undertaken once. This section provides method statements for the collation and analysis of the sorts of data that were available for the Berg River (Table 7.4), which included:

- 1:50 000 topographical maps and aerial images from the local Department of Surveys and Mapping;
- daily rainfall data from the South African Weather Service;
- daily discharge data from the Department of Water and Sanitation Hydrological Services website;
- community data from past (historical) biological surveys.

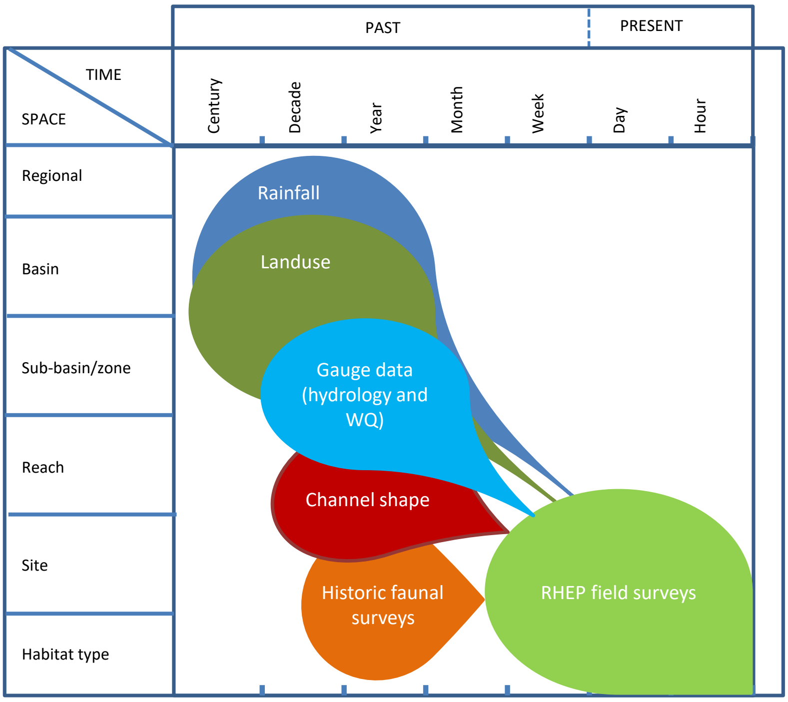

Figure 7.4 Temporal and spatial resolution for different types of data

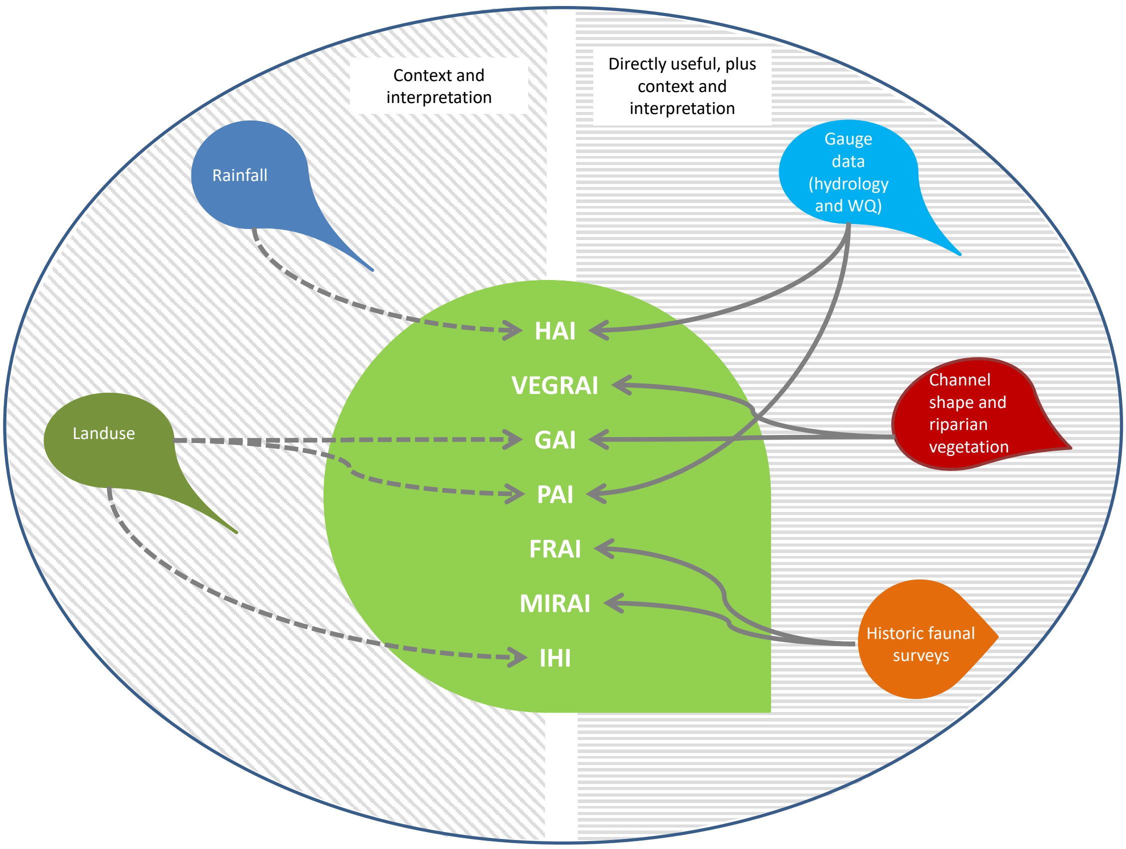

Figure 7.5 The potential use of historic data to inform the calculation of the indices required for REMP, and/or their analysis and interpretation

Table 7.4 Timeline of data available for the Berg River Basin

	1900	1910	1920	1930	1940	1950	1960	1970	1980	1990	2000	2010	2017
Topographic maps													
Aerial images													
Daily rainfall													
Discharge data													
Biological surveys													

Method statements based on the methods used in Chapters 3 - 6 are provided in the Appendices for the collation and assessment of data on land-use; rainfall and hydrology; channel change and riparian vegetation; and macroinvertebrates. These method statements are unlikely to cover all eventualities, as it is likely that the set of data that are available for another basin will differ either slightly or significantly from that available for the Berg River Basin. Nonetheless, the principles, approach and method statements outlined provide pragmatic examples of pragmatic and cost-efficient use of these sorts of data that may need to be adapted to other sorts of data sets.

The macroinvertebrate method statement deals only with FAMILY level identifications. This is based on the outcome of the assessment of the Berg River macroinvertebrate data sets in Chapter 6, which concluded that the expense and complications involved in the use of data at a lower taxonomic were not justified.

7.5 Discussion

Ecological research and monitoring is entering a new era of integration and collaboration as we meet the challenge of understanding the great complexity of biological systems (Thompson *et al*. 2001). As part of this, maximising information across a variety of disciplines and scales, made possible by developing new methods, technologies, and funding opportunities, provides information needed to manage and protect river systems.

This study demonstrated the value gained by investing in the collation and analysis of historical data in a simple, cost-effective manner and how this may be used to augment Reserve and monitoring data to provide a more robust definition of reference conditions for REMP indices and the interpretation of RQO monitoring data, and to provide a quantitative basis for what are otherwise highly subjective assessments. Information was gathered from wherever it could be found and then tested to see if it was useful in interpreting change in the Berg River. The study provided valuable lessons in making the most of data with irregular coverage, in patching data to ensure that it could be analysed, in combining data from different temporal and spatial scales, and in applying scales broader than those routinely considered in Reserve determinations or monitoring studies to augment the assessments required. There was more information available without charge than originally envisaged and so, with hindsight, the main investment was the time taken to collate and analyse it all. Furthermore, the insights gained greatly increased our understanding of the functioning of the Berg River ecosystem and the factors at play in its continuing management.

There were two main groups of changes in land-use; changes in agricultural land from one crop type to another, and then an increase in water resource use, mostly through an increase in the number of dams. Since changes in land-use also cause changes in flow, and changes in flow, in the case of the Berg River Basin, also resulted from a need to provide water to facilitate changes in land-use, these two components of change were inter-linked, often with knock-on effects for the river and associated aquatic habitats.

Pine plantations were removed from the upper foothills and dryland farming decreased in the lower foothills and the lowlands. The plantations were decommissioned and these areas were handed over to conservation authorities to recover naturally back to wilderness areas. In addition, there was an increase in the presence of woody alien plants between 2003 and 2006, which were subsequently cleared up to 2015, and there was a severe reduction in the area of the riparian zone at the same time. It is possible that this reduction is related to the indiscriminate clearing methods used to clear the alien trees, and in other cases there may be few indigenous species present from which unassisted natural recovery can take place, such as under stands of Eucalypts that release allelochemicals to hinder undergrowth (Ruwanza *et al.* 2015). In any case, the inability of the riparian zone to regenerate unassisted leaves these areas open to being cultivated or resulted in them being taken over by reeds and grasses, a phenomenon not uncommon following the clearing of woody alien trees but especially when nitrogen fixing trees were present (Jamu *et al.* 2003; Ruwanza *et al.* 2013).

There was also a shift from dryland farming to irrigated agriculture that took place (Reed and Kleynhans 2009; Stuckenberg *et al.* 2013). The dryland crop areas that were situated further away from the river were now left fallow and the areas closer to the rivers were changed over to cultivate irrigated orchards and vineyards. There was a concomitant increase in the number of farm dams all over the basin to support these new irrigated crops and also an increase in the regulation of flow along the Berg River through development of the Western Cape Water Supply System, an interlinked system of water transfers and abstractions from the various large dams in and outside of the Berg River Basin. These dams were built to release water for agricultural needs but also to supply potable water to various towns and cities in the Cape Metropole.

Agriculture is one of the biggest water users within the Western Cape, with 32% of water from WCWSS allocated for irrigation while 63% is used for domestic and industrial purposes within the City of Cape Town (Pegram and Baleta 2014). This has changed somewhat since the drought conditions of 2015, with agricultural demand for water supplies being severely restricted up to 60-80% of previous supply (Drought fact sheet 2017). The Western Cape has a Mediterranean climate where the growing season coincides with the low flow summer dry season so the high flow winter water gets stored for release during the dry irrigation period, using the river as a conduit to supply water. This increases the dry season low flows, a combination of an increase in discharge and the duration of the dry season, beyond what would naturally flow down the river. In the Western Cape, a decrease in macroinvertebrate taxon richness was reported due to increased dry season low flows after the Berg River inter-basin transfer scheme (Snaddon and Davies 1998). Although overall communities sampled had remained constant, deceases in SASS5 scores in the riffle and run habitats have been associated with elevated flows in the Berg River due to flow regulation and removal of riparian vegetation (Snaddon 2009).

Dams large and small have a different effect on flow during the wet season as they trap and store floods in the reservoir to be released during the dry season. In the Berg River Basin, the hydrological indicators assessed showed different changes in different parts of the basin due to flow regulation but there were some general trends that were common to most parts of the basin. In general, there was a decrease in the incidence of intra-annual floods overall, as there were fewer class 2 floods and a greater number of class 4 floods in the Berg River, as well as an overall decrease in the average discharge during the wet season.

A reduction in the number of intra-annual floods also reduces overall wetting of the braided river channels, the floodplain and the riparian area. This provides easier access to these areas that are targeted for cultivated crops due to their proximity to the river, for easy abstraction, and their nutrient-rich soils. Farmers fill in and drain floodplains and also bulldoze over braided channels to convert these areas for cultivated crops. Together, the changes in flow and the cultivation of floodplains and braided sections of the river channel simplify the complexity of aquatic habitat available to macroinvertebrates. The Berg River channel and tributaries had lost sinuosity, by being straightened, and there are now fewer channel braids and floodplains. Previously braided river reaches are now single-thread channels, with or without a floodplain, and previous floodplains are now either disconnected from the rivers by berms and so don't flood anymore, or have been drained, infilled and cultivated. All of these changes have led to an overall decrease in the river channel and riparian area. Floodplains and braided channels slow floods down and absorb flood waters. This was evident in an increase in the peak magnitude and duration of the larger floods, the effects of which increased downstream.

The presence of dams and their regulation of flow also changes the volume and timing sediment transport. According to Kummu and Varies (2007), dams around the world have a high sediment trapping efficiency, up to 80%, and about 1% of existing storage volume is lost each year (Kummu and Varis 2007).

Changes in sediment transport up- and downstream of dams and in response to changes in flow between the wet and dry season, also changes the degree of erosion and sediment deposition that occurs, which changes the distribution of sediment in the channel and therefore quality of habitat for macroinvertebrates. The changes in land-use, flow and sediment transport, largely driven by changes in agricultural water needs, all have knock-on effects for water quality and quality of aquatic habitats, all of which influence the distribution and abundance of macroinvertebrates in the river and tributaries. This was evident in the weaker correlations between changes in communities of aquatic invertebrates and the hydrology indicators, when compared to that between invertebrates, land-use and changes in the habitats of the river channel.

The main changes seen in the communities of macroinvertebrates were a reduction in the abundance of organisms that are sensitive to pollution and changes in water quality since the first detailed study in the 1950s are well documented elsewhere (Coetzer 1978; Dallas 1992; Ractliffe *et al.* 2007), but also a reduction in the abundance of organisms that are sensitive to changes in habitat quality, in response to changes in flow and changes in the distribution of sediments along the river. There was also an increase in organisms associated with marginal vegetation and slow flowing areas as well as others associated with slow flows, fine sediments detritus. The organisms in the upper foothills were generally more sensitive to pollution and poor water quality than those in the lowlands, which were hardier.

Overall, considering all variables across the basin, the main drivers of changes in the communities of macroinvertebrates were the removal of the plantations and associated increase in undeveloped lands, reduced discharge in the wet season and the increase in volume of the dry season, and the reduction in the proportion of the river channel that was braided.

The history of an ecosystem's structure and functioning is vital for understanding how present conditions came about (Turner 2005), how ecosystems function, and for defining reference conditions (Newson 2008). As such, collating and analysing the past is crucial to ecologically-sound management (Bis *et al.* 2000; Rhemtulla and Mladenoff 2007). The distribution of organisms at large and small scales is influenced by the natural and anthropogenic histories (Turner 1989).

Taking the time to understand and analyse past activities in our river basins and how these have affected the ecological conditions of the river that drain them is an investment that should be made to better manage our inland aquatic ecosystems. It is especially important as the focus of the DWS shifts from setting Reserves to implementing and monitoring RQOs, and in light of the shortage of budget, skills and access to robust data sets needed to ensure comprehensive and scientifically-defendable monitoring and reporting of successes and failures. It would be naïve to presume that implementation of the Ecological Reserve and its efficacy in meeting the agreed RQOs will not at some point be challenged in a court of law, which means that it is sensible to maximise the use of ***<u>all</u>*** available data to support monitoring, reporting and the conclusions drawn in that regard.

The main sources of data on whether or not the Ecological Reserve is being correctly implemented in a basin, and its efficacy in sustaining the RQOs for that basin, are intended to be the REMP and the RQO monitoring programmes (http://www.dwa.gov.za/iwqs/rhp/rhp_background.aspx). For the most part, these focus on sites established in Reserve determination studies, and use the hydrological and ecological information developed in those studies as benchmarks for ongoing monitoring. While this pragmatic approach is understandable, it is important to recognise that the data collection and analyses done for the Reserve determination studies were not aimed at providing baselines for ongoing monitoring, and as such the data from those studies are likely to need augmentation to achieve defensible definitions of baseline/reference conditions on which to base monitoring. This is particularly so for the REMP indices, which are assessed according to change from ‘reference’ conditions, and thus presumably depend on a robust definition of reference conditions.

This study demonstrated the value gained by investing in the collation and analysis of historical data in a simple, cost-effective manner and how this may be used to augment Reserve and monitoring data to provide a more robust definition of reference conditions for REMP indices and the interpretation of RQO monitoring data, and to provide a quantitative basis for what are otherwise highly subjective assessments. Information was gathered from wherever it could be found and then tested to see if it was useful in interpreting change in the Berg River. The study provided valuable lessons in making the most of data with irregular coverage, in patching data to ensure that it could be analysed, in combining data from different temporal and spatial scales, and in applying scales broader than those routinely considered in Reserve determinations or monitoring studies to augment the assessments required. There was more information available without charge than originally envisaged and so, with hindsight, the main investment was the time taken to collate and analyse it all. Therefore widespread and systematic incorporation of large scale long-term data into monitoring and interpreting the implementation, and adaptive management, of the Ecological Reserve makes scientific and financial sense.

8 Conclusion

Five linkages between the lotic landscape and the river ecosystem were described in the introduction to this dissertation. These were: land-use, hydrology, channel shape, riparian vegetation and the biota that inhabit the river ecosystem. Historical changes in these were addressed by testing eight hypotheses, and informed by a review of the international literature that focussed on the structure and functioning of river ecosystems with a particular emphasis on the river flow, sediment transport, nutrients, organic matter and biota in different riverine habitats. Of the eight hypotheses, four were supported, two were rejected and two were inconclusive. In summary, the four hypotheses supported were: that the rate of change in land-use has accelerated over time; that land-use and water-resource developments in the Berg River Basin have affected the volume and distribution of flows in the Berg River; that, in general, the changes that occur in river planform as a result of development will tend towards narrower systems with less habitat diversity, and; that macroinvertebrate assemblages in the Berg River have changed over time, with tolerant taxa becoming more dominant. The two hypotheses that were rejected were: that changes in land-use are progressive and; that the effects of land-use change on the flow regime can be isolated from changes due to large impoundments and water-resource schemes (e.g. Theewaterskloof-Berg Scheme, Berg River Dam, Voelvlei Dam and Misverstand Dam). The remaining two hypotheses, viz.: that different land-uses affect the riparian area and river channel structure in different ways, and; that changes in macroinvertebrate community structure are linked more closely with changes in habitat and water quality than changes in flow, were inconclusive.

That land-use and water-resource developments in the Berg River Basin have affected the volume and distribution of flows in the Berg River was supported. The data collected supported the hypotheses that accelerated change in land-use and water resource developments in the Berg River Basin have affected the volume and distribution of flows in the Berg River. These effects were more strongly felt lower down in the basin as the incremental area over which flows were regulated is larger. Aspects affected included an increase in the dry season flows, shown by a longer dry season with a greater average discharge, and a reduction in the number of intra-annual floods that contributed to lower flow volumes in the wet season. The effects of changes in land-use and water resource developments are well described in the international literature (Ward and Stanford 1979; Petts 1987; Bunn and Arthington 2002; King *et al*. 2003a). Changes in land-use have been shown to affect rivers and other aquatic ecosystems (Lessard and Hayes 2003).

Literature has communicated the various impacts of water resource developments on flow regime, channel structure, sediment production and transport, water quality and others; all of which have knock-on effects to aquatic flora and fauna assemblage structuring and functioning (e.g. Ward and Stanford 1979; Petts 1987; Ligon *et al.* 1995; Poff *et al.* 1997; Pringle *et al.* 2000; Paul and Meyer 2001; Bunn and Arthington 2002; King *et al*. 2003a; Lessard and Hayes 2003; Butler and Davies 2004). In America, a reduction in annual peak discharges by 67% (in some individual cases up to 90%) and decrease in daily discharges by 64% were reported for rivers with dams when compared to those without dams (Graf 2006). Dams also trap moving sediment causing downstream channel and bank erosion, and a concomitant degradation of in-channel and riparian habitats (William and Wolman 1984; Petts 1985).

In addition to other water development schemes on the Berg River, an inter-basin water transfer from the neighbouring Riviersonderend River through the Theewaterskloof Dam operated since late 1980. Transfer of water between river basins has also been pointed as a major disruptor to the flow and ecosystems of both rivers involved (Davies *et al*. 1992; Snaddon *et al.* 2000); i.e., inter-basin transfers alter flow regime, changes in water quality, loss of of biogeographical integrity, the loss of endemic biotas, and introduction of alien (Davies *et al.* 1992). Decrease in macroinvertebrate taxon richness, loss of sensitive taxa and an increase in collectors-predators of the Berg River below the outlet has been reported by Snaddon and Davies (1998). After the Orange-Fish River inter-basin transfers, an increase in the mean annual runoff was at the Fish River and a significant change in macroinvertebrate communities of the riffle habitat were accompanied by loss of diversity as only 33% of taxa identified were common to those of before the water transfers (O'Keeffe and De Moor 1988).

The other supported hypotheses were that development in the river basin changes channel planform and simplified the complexity and structure of river ecosystems, which decreased the diversity of habitat in the river and contributed to a change in the aquatic biota. The data showed that the number and abundance of aquatic macroinvertebrates tolerant to reduced habitat quality increased. The river changed over time exhibiting reductions in channel sinuosity, channel braiding, the area of the river channel and riparian zone, the area of floodplains and of sand bars. The riparian area changed as trees and shrubs were replaced by reeds and grasses, or by woody alien trees that were subsequently cleared and then replaced by reeds and grasses. These kinds of changes are reported globally due to agricultural development in river basins (Beschta *et al.* 1987; Poff and Hart 2002). In many ways the structure and function of river ecosystems has been adversely impacted by dams (Poff and Hart 2002). Change in water and sediment transport and distribution will have knock on effects on the structure and dynamics of aquatic and riparian habitats and biota (Beschta *et al.* 1987).

Dams also create a barrier for biota movement and nutrient distribution, which separates the river communities (such as fish and macroinvertebrates) upstream and downstream of the dam and reservoir. Dams also change downstream water temperature (Poff and Hart 2002), which can influence rates of spawning, egg development, growth success, metabolism, species competition (Beschta *et al.* 1987; Paller and Saul 1996; Dallas 2008; Dallas 2009b). In the Columbia River, Pacific Northwest, increased water temperature below dams encouraged anadromous salmonids fish to migrate to the cooler mouth (Beschta *et al.* 1987). The removal of forest vegetation along channels has been associated with significant change increase in water temperatures (Harding *et al.* 1999) as they work as buffer strips.

The hypotheses that land-use changed progressively was rejected since changes in land-use were not unidirectional. Changes in extent of different land-use classes (cultivated land, shrubs and woodlands) within the Likalanga Basin in Malawi, between 1984 and 2013 had been unidirectional. For instance the overall extent of cultivated and grazing land had increased from 350 km^2 in 1984 to 502 km^2 in 2013; however this was not a continuous progressive increase over time, as the area of this class was 350 km^2 in 1984, which had decreased in 1994 (289 km^2) then increased in 2005 (478 km^2) followed by another increase by 2013 (502 km^2) (Pullanikkatil *et al.* 2016). The extent of some land-use types increased over time, such as urban development, others decreased over time, such as agricultural land (plantations), and others increased first only to decrease again, such as agricultural land (dryland farming). These types of results are understandable when one considers how farmers in this river basin have had to respond to prevailing changes in climate over successive wet and dry years, including droughts, the increase in the tourism industry and the move towards lifestyle farms, conservancies and private nature reserves.

There was insufficient evidence to support or reject the hypotheses that different land-uses affected river channel structure and the riparian area in different ways. It was not possible to separate out differences in the areas delineated from the natural overriding biophysical factors due to longitudinal positioning. Aquatic ecosystems structural and functional characteristics are influenced by their position within the river basin or overall state of the physical system (Vannote *et al.* 1980; Allan and Johnson 1997). For instance, the River Continuum Concept (Vannote *et al.* 1980), which explains longitudinal connectivity in rivers, recognizes that they function differently between the upper, middle and lower reaches of the river system (Rodriguez-Iturbe and Rinaldo 2001; Finlayson and McMahon 2004). Also, land-use differed across the basin and so the effects of this varied spatially as well as over time, with changes in the upper foothills being driven by the conversion of pine plantations to naturally vegetated conservation areas. Changes in the lowlands were being driven by dryland crops being given over to fallow fields and in the upper and lower foothills there was an increase in irrigated orchards and vineyards along the rivers and on the floodplains. The difficulty of multi-disciplinary, large scale studies separating out co-related variables that influence rivers ecosystems has been expressed (McDonnell 2008). For large scale studies that have different land-uses it is be difficult to discern and isolate the hydrological effects of land cover change (Costa *et al.* 2003). In order to link changes in hydrology to those of land-use, a long enough data set that has considerable changes in land-use between them is required (Costa *et al.* 2003).

It was also not clear that flow was responsible for the extensive change in the composition of the macroinvertebrate community over and above changes in water quality and habitat, and it was also difficult to distinguish changes in community composition in response to natural biophysical changes longitudinally down the river, from those in response to anthropogenic change in different parts of the river basin. It is all but impossible to separate out natural change from anthropogenic change especially in basins where the level of agricultural development and regulation of flow are as high as they are in the Berg River Basin.

A meta-analysis showed the main drivers causing change in macroinvertebrate communities were a shift from dryland agriculture to fallow lands and an increase in water demand to supply the increase in irrigated crops along the rivers, which led to a greater number of farm dams being built and greater regulation of flow down the Berg River through the various impoundments and water resource systems. This elevated and extended the duration of flows, the dry season beyond what would normally occur and reduced the number of intra-annual floods taking place, which contributed to a reduction in average discharge in the wet season.

Finally, the two main management issues addresses in this dissertation are basin-wide links between changes in land-use and flow, and more localised links between changes in aquatic habitat diversity and biota. Taking account of historical trends in the baseline information against which change is measured, as done in this dissertation, allows us to better quantify the extent to which change takes place in river basins, also to identify sources of change. This sets water resource management in a better position to be able to adapt to unforeseen change more realistically.

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10 Appendices

Appendix A. CHAPTER 3: ADDITIONAL INFORMATION

Appendix Table 1 Longitudinal gradient and altitude associated with each river zone

Geomorphological zones (gradient class)	Cumulative distance (m)	Altitude (m)	Gradient range
Mountain headwater stream (>0.1)	2504	540	0.135 - 0.454
Mountain stream (0.04-0.099)	2710	520	0.068 - 0.097
Transitional (0.02-0.039)	4442	400	0.028 - 0.029
Upper foothills (05-0.019)	6457	340	06 - 0.016
Lower foothills (01-05)	22914	160	01 - 03
Lowland (001-0009)	86381	60	001 - 003

The data layers were grouped into 10-year periods to maximize use of available maps (Appendix Table 2). In this study, these 10-year periods are treated as a single point in time. The intention was to investigate change for the time periods 1900-1930, 1930-1959, 1960-1969, 1970-1979, 1980-1989, 1990-1999, 2000-2009 and 2010-2015, however, there were many missing maps so the periods were revised and adjusted accordingly. For example, 1930-1959 had only 10 maps. With 15 maps missing there could not be proper comparisons for this period with the other ones. The periods eventually used were 1955-1965, 1976-1985, 1996-2005 and 2006-2015 (Table 3.4).

Appendix Table 2 An inventory of topographical maps sourced for the study

Map index	Periods						
	1930-1959	1960-1969	1970-1979	1980-1989	1990-1999	2000-2009	2010-2015
3217 db_dd	1942	1964		1980	1998	2003	
3218 ca_cc		1965		1981	1998	2003	2010
3218 cb_ca		1964		1986		2003	
3218 cd		1965		1982		2003	2010
3218 da		1964		1986		2003	
3218 dc		1961	1975			2003	2010
3218 dd		1961	1975			2003	
3219 cc		1963		1986		2003	2010
3317_3318 aa	1942	1966		1981	1998		2010
3318 ab		1967		1980		2003	2010
3318 ac							
3318 ad	1942	1966		1981	1999		2010
3318 ba	1943	1966		1980		2000	2010
3318 bb	1942	1963		1981		2000	2010
3318 bc	1942	1966	1979			2000	2010
3318 bd		1963	1978	1988	1997	2000	2010
3318 da							
3318 db	1941	1963	1977		1997-1992	2000	2010
3318 dd	1935/1959				1992	2000	2010
3319 aa			1971		1997		2010
3319 ac	1945		1971		1997		2010
3319 ad							

Map index	Periods						
	1930-1959	1960-1969	1970-1979	1980-1989	1990-1999	2000-2009	2010-2015
3319 ca	1958		1979		1997		
3319 cc		1962	1977		1997		2010
3419 aa		1963	1979		1997		2010

Appendix Table 3 Annual incremental values of patched maps for each agricultural land use

Land-use classes	3319ac	3319aa	3318dd
Dryland farming	0.31	0.10	-0.51
Orchards and vineyards	0.97	1.91	1.98
Plantations	0.10	0	-0.27

A.1. LAND-USE PER PERIOD

In 1955-1965 (Appendix Table 4):

- ±80.6% of the basin was under agriculture, comprising ±7219.1 km^2. Dryland farming, orchards and vineyards, and plantation were present in all the sub-basins with the exception of the estuary, which had only dryland farming. The biggest extent of agricultural lands was in the area draining into the lowland (±58.3%), estuary (±12.8%) and lower foothills of the Berg River (±9.2%), of which dryland farming comprised ±55.0%, ±12.8% and ±5.6%, respectively. By contrast, agriculture in the upper foothills was dominated by orchards and vineyards (±0.3% of total basin area); plantation and dryland farming covered combined were <1% both at 0.01%.
- Urban areas made up to ±0.5% of the total basin, towns covered ±0.4% and townships were ±0.1%. The lower foothills (±0.2%) and lowlands (±0.2%) had the highest proportion of urban developed land.
- ±18.89% of basin area was undeveloped land, with the largest undeveloped areas in the estuary (±9.6%), lowland (±4.2%) and lower foothills (±3.4%) while the upper had the smallest (±1.6%).
- Outside of the urban areas, there were ±1654 buildings, most of which were farms (±1590) and townships (±28). The number of towns and industrial buildings were the same (±18). The lowland (±881), lower foothills (±558) and estuary (±172) had the highest number of buildings, while the upper foothills had the least (±43).
- There were ±1325 waterbodies in the whole basin, most of which were farm dams (±836) and non-perennial pans (±335). The highest number of farm dams was found in the lowlands (±943), followed by the lower foothills (±243) and estuary (±132). The upper foothills had only seven waterbodies, all of which were dams.

Appendix Table 4 The area/number of different land-use types (1955-1965)

Land-use class		Upper foothills		Lower foothills		Lowland reaches		Estuary	
		km²	%	km²	%	km²	%	km²	%
Area									
Agricultural	Dryland farming	0.7	0.01	500.4	5.6	4928.2	55.0	1142.5	12.8
	Orchards and vineyards	26.7	0.3	280.7	3.1	210.1	2.3	0	0
	Plantations	1.3	0.01	42.0	0.5	86.4	1.0	0	0
	Total agricultural	**28.8**	**0.3**	**823.0**	**9.2**	**5224.7**	**58.3**	**1142.5**	**12.8**
Urban	Towns	1.4	0.02	13.5	0.2	15.4	0.2	7.2	0.1
	Townships	0.6	0.01	4.3	0.05	1.9	0.02	2.1	0.02
	Total urban	**2.0**	**0.02**	**17.8**	**0.2**	**17.3**	**0.2**	**9.3**	**0.1**
Undeveloped land		142.1	1.6	306.7	3.4	379.8	4.2	863.9	9.6
Counts									
Buildings outside of urban areas	Farms	38		535		858		159	
	Industrial buildings	0		4		8		6	
	Towns	1		5		8		4	
	Townships	4		14		7		3	
	Total buildings	**43**		**558**		**881**		**172**	
Water bodies	Dams	7		237		588		4	
	Dry pans	0		1		112		7	
	Non-perennial pans	0		4		231		100	
	Perennial pans	0		1		12		21	
	Water bodies total	**7**		**243**		**943**		**132**	

In 1976-1985 (Appendix Table 5):

- ±85.4% of the basin was under agriculture, comprising ±7466.3 km^2; dryland farming, orchards and vineyards, and plantation were present in all the sub-basins with the exception of the estuary, which had only dryland farming. The biggest extent of agricultural lands was in the lowlands (±59.9%), estuary (±12.4%) and lower foothills (±9.9%^{2}); dominated by dryland farming ±56.9%, ±12.4%, and ±5.1%, respectively. The upper foothills were dominated by plantations (0.71%); orchards and vineyards (±0.4%) with very little dryland farming (±0.01%).
- Urban areas comprised ±0.6% of the total basin, towns covered ±0.4% and townships ±0.14%. The lower foothills (±0.2%), estuary (±0.19%) and lowlands (±0.17%) had the largest urban areas, while the upper foothills (±0.02%) had the lowest.
- ±15.9% of the basin was undeveloped land, the highest proportion was found in estuary (±9.9%), both lowlands and lower foothills (±2.7%) while the upper Berg had the lowest (±0.7%).
- There were ±1446 buildings outside of urban areas, most of which were farms (±1398) and towns (±21) with a few industrial buildings (±17) and townships (±10). The lowlands (±802), lower foothills (±314) and estuary (±274) had the highest number of buildings outside of established urban areas, while the upper foothills and mountain stream (±56) had the least.
- There were ±1490 waterbodies, most of which were farm dams (±1015) and non-perennial pans (±369). The highest number of waterbodies was found in the lowlands (±837) followed by the lower foothills (±467), estuary (±139) and upper foothills (±47).

Appendix Table 5 The area/number of different land-use types (1976-1985)

Land-use class		Upper foothills		Lower foothills		Lowland reaches		Estuary	
		km²	%	km²	%	km²	%	km²	%
Area									
Agricultural	Dryland farming	0.54	0.01	459.1	5.1	5098.5	56.9	1110.7	12.4
	Orchards and vineyards	37.5	0.42	389.1	4.3	223.7	2.5	0	0
	Plantations	63.3	0.71	40.0	0.45	44.0	0.49	0	0
	Total agricultural	**101.3**	**1.1**	**888.2**	**9.9**	**5366.2**	**59.9**	**1110.7**	**12.4**
Urban	Towns	1.47	0.02	11.2	0.13	12.3	0.14	15.5	0.17
	Townships	0.05	0	8.5	0.09	2.9	0.03	1.53	0.02
	Total urban	1.52	0.02	19.7	0.22	15.3	0.17	17.1	0.19
Undeveloped land		70.0	0.78	239.6	2.7	240.4	2.7	888.0	9.9
Counts									
Buildings outside of urban areas	Farms	54		303		785		256	
	Industrial buildings	0		3		3		11	
	Towns	2		4		8		7	
	Townships	0		4		6		0	
	Total buildings	**56**		**314**		**802**		**274**	
Water bodies	Dams	47		462		499		7	
	Dry pans	0		0		6		37	
	Non-perennial pans	0		1		311		57	
	Perennial pans	0		4		21		38	
	Water bodies total	**47**		**467**		**837**		**139**	

In 1996-2005 (Appendix Table 6):

- ±77.8% (±6972.4 km^2) of the basin under agriculture; dryland farming, orchards and vineyards, and plantations were present in all the sub-basins except for the estuary, which did not have plantations. The largest extent of agricultural lands was found in the lowlands (±57.3%), estuary (±11%) and lower foothills (±8.9%), which were dominated by dryland farming, ±54.1%, ±11%, and 4.5%, respectively. The upper foothills were dominated by orchards and vineyards (±0.4%) and plantations (±0.3%), with very little dryland farming (±0.01%).
- Urban areas comprised ±1.1% of the total basin; towns covered ±0.86% and townships ±0.2%. The lower foothills (±0.5%) and estuary (±0.3%) had the highest proportion of built up land.
- ±21.1% of basin area was undeveloped land (±1885.6 km^2), with the largest undeveloped areas in the estuary (±11.2%), lowlands (±5.2%) and lower foothills (±3.5%) while the upper reaches had the smallest (±1.2%).
- There were ±1542 buildings outside of urban areas, most of which were farms (±1387) and townships (±118). Most non-urban buildings were located in the lowlands (±846) and lower foothills (±479), while the estuary (±136) and upper foothills and mountain stream (±81) had the least.
- There were ±3077 waterbodies, most of which were farm dams (±2439) and non-perennial pans (±569). The highest number of waterbodies was found in the lower berg (±1874) this was followed by the middle berg (880), estuary (199), while the upper berg had ±124.

Appendix Table 6 The area/number of different land-use types (1996-2005)

Land-use class		Upper foothills		Lower foothills		Lowland reaches		Estuary	
		km²	%	km²	%	km²	%	km²	%
Area									
Agricultural	Dryland farming	1.2	0.01	400.6	4.5	4841.8	54.1	981.1	11.0
	Orchards and vineyards	31.5	0.4	368.8	4.1	246.3	2.7	1.1	0.01
	Plantations	29.4	0.3	26.6	0.3	44.1	0.5	0.0	0.0
	Total agricultural	**62.0**	**0.7**	**796.0**	**8.9**	**5132.2**	**57.3**	**982.2**	**11.0**
Urban	Towns	1.7	0.02	30.3	0.3	19.6	0.2	25.5	0.3
	Townships	0.0	0.0	11.8	0.1	5.4	0.06	5.6	0.06
	Total urban	**1.7**	**0.02**	**42.1**	**0.5**	**25.0**	**0.28**	**31.1**	**0.3**
Undeveloped land		109.1	1.2	309.4	3.5	464.7	5.2	1002.5	11.2
Counts									
Buildings outside of urban areas	Farms	75		442		769		101	
	Industrial buildings	0		3		7		6	
	Towns	2		4		11		4	
	Townships	4		30		59		25	
	Total buildings	**81**		**479**		**846**		**136**	
Water bodies	Dams	125		876		1426		13	
	Dry pans	0		0		0		0	
	Non-perennial pans	0		4		443		122	
	Perennial pans	0		0		5		64	
	Water bodies total	**124**		**880**		**1874**		**199**	

In 2006-2015 (Appendix Table 7):

- Land under agriculture covered 71.1% (±6365.4 km^2) of the basin; all land-use classes were present across all sub-basins except for plantation which was not present on the upper foothills and estuary. The largest area of agricultural lands was found in the lowlands (±53.4%), estuary (±10.4%) and lower foothills (±7%), which were all dominated by dryland farming, ±50.2%, ±10.4% and ±3.9%, respectively. The upper foothills were dominated by orchards and vineyards ±0.2% with very little dryland farming (±0.01%) and no plantations.
- Urban areas made up to 1.2% of the total basin, towns covered a total of ±0.8% and townships were ±0.3%. The estuary (±0.6%) and lower foothills (±0.5%) had the highest proportion of land that is built-up.
- ±27.7% of basin area was undeveloped land (±2481.1 km^2), the largest areas of undeveloped land were in the estuary (±11.5%), lowlands (±9.3%), lower foothills (±5.3%), while the upper foothills had the smallest (±1.6%).
- There were a total of ±1507 buildings, of which most of them were farms (±1084) and townships (±369). Most buildings were located on the lowlands (±787) and lower foothills (±438) while the estuary (±157) and upper foothills (±125) had the least.
- There were a total of ±3052 waterbodies, most of which were dams (±2418) and non-perennial pans (±566). The highest number of waterbodies was found in the lowlands (1840) this was followed by the lower foothills (±890), estuary (±197), while the upper foothills had ±125.

Appendix Table 7 The area/number of different land-use types (2006-2015)

Land-use class		Upper foothills		Lower foothills		Lowland reaches		Estuary	
		km^2	%	km^2	%	km^2	%	km^2	%
Area									
Agricultural	Dryland farming	1.2	0.01	352.8	3.9	4493.6	50.2	933.8	10.4
	Orchards and vineyards	21.9	0.2	263.7	2.9	262.3	2.9	1.35	0.02
	Plantations	0	0	9.9	0.1	25.0	0.28	0	0
	Total agricultural	**23.1**	**0.3**	**626.4**	**7.0**	**4780.8**	**53.4**	**935.1**	**10.4**
Urban	Towns	1.5	0.02	40.4	0.5	2.5	0.0	35.7	0.4
	Townships	1.7	0.02	7.9	0.1	7.6	0.1	14.1	0.2
	Total urban	**3.2**	**0.04**	**48.3**	**0.5**	**10.1**	**0.1**	**49.7**	**0.6**
Undeveloped land		146.5	1.6	472.8	5.3	830.9	9.3	1030.8	11.5
Counts									
Buildings outside of urban areas	Farms	121		399		501		63	
	Industrial buildings	0		8		9		6	
	Towns	1		7		15		8	
	Townships	3		24		262		80	
	Total buildings	**125**		**438**		**787**		**157**	
Water bodies	Dams	125		885		1395		13	
	Dry pans	0		0		0		0	
	Non-perennial pans	0		4		440		122	
	Perennial pans	0		1		5		62	
	Water bodies total	**125**		**890**		**1840**		**197**	

Appendix Table 8 Counts and percentage of total basin counts of water bodies across the basin overall, and changes per time period

Sub-basin	Category	Land-use classs	1955-1965		Difference		1976-1985		Difference		1996-2005		Difference		2006-2015	
			Count	% of basin	Count	% increase / decrease basin	Count	% of basin	Count	% increase / decrbasin	Count	% of basin	Count	% increase / decrease basin	Count	% of basin
Upper foothills	Buildings outside of urban areas	Farms	38	2.39	16	1.01	54	3.86	21	1.50	75	5.41	46	3.32	121	11.16
		Industrial buildings	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
		Towns	1	5.56	1	5.56	2	9.52	0	0.00	2	9.52	-1	-4.76	1	3.23
		Townships	4	14.29	-4	-14.29	0	0.00	4	40.00	4	3.39	-1	-0.85	3	0.81
		Total buildings	43	2.60	13	0.79	56	3.87	25	1.73	81	5.25	44	2.85	125	8.29
	Water bodies	Dams	7	0.84	40	4.78	47	4.63	78	7.68	125	5.12	0	0.00	125	5.17
		Dry pans	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
		Non-perennial pans	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
		Perennial pans	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
		Water bodies total	7	0.53	40	3.02	47	3.15	77	5.17	124	4.03	1	0.03	125	4.10
Lower foothills	Buildings outside of urban areas	Farms	535	33.65	-232	-14.59	303	21.67	139	9.94	442	31.87	-43	-3.10	399	36.81
		Industrial buildings	4	22.22	-1	-5.56	3	17.65	0	0.00	3	18.75	5	31.25	8	34.78
		Towns	5	27.78	-1	-5.56	4	19.05	0	0.00	4	19.05	3	14.29	7	22.58
		Townships	14	50.00	-10	-35.71	4	40.00	26	260.00	30	25.42	-6	-5.08	24	6.50
		Total buildings	558	33.74	-244	-14.75	314	21.72	165	11.41	479	31.06	-41	-2.66	438	29.06
	Water bodies	Dams	237	28.35	225	26.91	462	45.52	414	40.79	876	35.90	9	0.37	885	36.60

		Dry pans	1	0.83	-1	-0.83	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
		Non-perennial pans	4	1.19	-3	-0.90	1	0.27	3	0.81	4	0.70	0	0.00	4	0.71
		Perennial pans	1	2.94	3	8.82	4	6.35	-4	-6.35	0	0.00	1	1.45	1	1.47
		Water bodies total	243	18.34	224	16.91	467	31.34	413	27.72	880	28.60	10	0.32	890	29.16
Lowland reaches	Buildings outside of urban areas	Farms	858	53.96	-73	-4.59	785	56.15	-16	-1.14	769	55.44	-268	-19.32	501	46.22
		Industrial buildings	8	44.44	-5	-27.78	3	17.65	4	23.53	7	43.75	2	12.50	9	39.13
		Towns	8	44.44	0	0.00	8	38.10	3	14.29	11	52.38	4	19.05	15	48.39
		Townships	7	25.00	-1	-3.57	6	60.00	53	530.00	59	50.00	203	172.03	262	71.00
		Total buildings	881	53.26	-79	-4.78	802	55.46	44	3.04	846	54.86	-59	-3.83	787	52.22
	Water bodies	Dams	588	70.33	-89	-10.65	499	49.16	927	91.33	1426	58.44	-31	-1.27	1395	57.69
		Dry pans	112	93.33	-106	-88.33	6	13.95	-6	-13.95	0	0.00	0	0.00	0	0.00
		Non-perennial pans	231	68.96	80	23.88	311	84.28	132	35.77	443	77.86	-3	-0.53	440	77.74
		Perennial pans	12	35.29	9	26.47	21	33.33	-16	-25.40	5	7.25	0	0.00	5	7.35
		Water bodies total	943	71.17	-106	-8.00	837	56.17	1037	69.60	1874	60.90	-34	-1.10	1840	60.29
Estuary	Buildings outside of urban areas	Farms	159	10.00	97	6.10	256	18.31	-155	-11.09	101	7.28	-38	-2.74	63	5.81
		Industrial buildings	6	33.33	5	27.78	11	64.71	-5	-29.41	6	37.50	0	0.00	6	26.09
		Towns	4	22.22	3	16.67	7	33.33	-3	-14.29	4	19.05	4	19.05	8	25.81
		Townships	3	10.71	-3	-10.71	0	0.00	25	250.00	25	21.19	55	46.61	80	21.68
		Total buildings	172	10.40	102	6.17	274	18.95	-138	-9.54	136	8.82	21	1.36	157	10.42
	Water bodies	Dams	4	0.48	3	0.36	7	0.69	6	0.59	13	0.53	0	0.00	13	0.54
		Dry pans	7	5.83	30	25.00	37	86.05	-37	-86.05	0	0.00	0	0.00	0	0.00
		Non-perennial pans	100	29.85	-43	-12.84	57	15.45	65	17.62	122	21.44	0	0.00	122	21.55
		Perennial pans	21	61.76	17	50.00	38	60.32	26	41.27	64	92.75	-2	-2.90	62	91.18
		Water bodies total	132	9.96	7	0.53	139	9.33	60	4.03	199	6.47	-2	-0.06	197	6.45
Totals	Buildings outside of urban areas	Farms	1590	100.00	-192	-12.08	1398	100.00	-11	-0.79	1387	100.00	-303	-21.85	1084	100.00
		Industrial	18	100.00	-1	-5.56	17	100.00	-1	-5.88	16	100.00	7	43.75	23	100.00

		buildings														
		Towns	18	100.00	3	16.67	21	100.00	0	0.00	21	100.00	10	47.62	31	100.00
		Townships	28	100.00	-18	-64.29	10	100.00	108	1080.00	118	100.00	251	212.71	369	100.00
		Total buildings	1654	100.00	-208	-12.58	1446	100.00	96	6.64	1542	100.00	-35	-2.27	1507	100.00
	Water bodies	Dams	836	100.00	179	21.41	1015	100.00	1425	140.39	2440	100.00	-22	-0.90	2418	100.00
		Dry pans	120	100.00	-77	-64.17	43	100.00	-43	-100.00	0	0.00	0	0.00	0	0.00
		Non-perennial pans	335	100.00	34	10.15	369	100.00	200	54.20	569	100.00	-3	-0.53	566	100.00
		Perennial pans	34	100.00	29	85.29	63	100.00	6	9.52	69	100.00	-1	-1.45	68	100.00
		Water bodies total	1325	100.00	165	12.45	1490	100.00	1587	106.51	3077	100.00	-25	-0.81	3052	100.00

Appendix B. CHAPTER 4: PATCHING HYDROLOGICAL RECORDS

Three main methods were used for patching the daily hydrological records, depending on the size of the gap and the availability of reference flow gauge or rainfall data from which to estimate monthly volumes and distributions of daily flow. In brief, the three methods were:

1. Method 1: (for data gaps of less than a month; reference gauge available). The volume and distribution of flow in the gauge with missing data were estimated by comparison with those from the gauge used for patching, based on the relative sizes of their MARs).
2. Method 2: (for data gaps of more than a month; reference gauge available). The missing volume was estimated using regressions developed for the gauge with missing data, and either another flow gauge, or a rainfall gauge (or an average of the two estimates).
3. Method 3: (for data gaps of more than a month; reference gauge not available). The missing volume was estimated using a regression relationship with rainfall, and apportioned using an "average" distribution of flow for wet or dry months for that flow gauge.

In the description below, the gauge being patched is called Gauge A, and the gauge used for patching is called Gauge B.

The worked examples of each of the methods use G1H004, which had the longest flow record and many gaps.

B.1. METHOD 1: DATA GAPS OF LESS THAN A MONTH AND A REFERENCE GAUGE AVAILABLE.

The volumes of flows contained in the days with data missing in Gauge A were estimated by comparison with the corresponding days in Gauge B. The volume of each of the missing days was then be apportioned to match the pattern of Gauge B. The steps were as follows:

a. For Gauge B, find the total volume for that month. This volume = B.

b. For Gauge B, find the volume contained in the days which are <u>not</u> missing in Gauge A (and by default the volume in Gauge B of the days which <u>are</u> missing in Gauge A).
The volume of not missing days in Gauge B = x.

c. For Gauge A, find the volume contained in the days which are <u>not</u> missing.
The volume of not missing days in Gauge A = y.

d. For Gauge B calculate the ratio of total volume for the month to the volume of the "not missing days". i.e. find B/x.

e. Find the total month's volume, A, for Gauge A by multiplying the volume of not missing days (y) by the ratio of total volume to not missing days volume (B/x): i.e.:
Total volume for the month in Gauge A = $A = y \times B/x$.
Subtracting the "not missing days" volume (y) from the total volume (A), will give the volume of the missing days (z). i.e. $z = A - y$

f. For Gauge B, find the portion of total volume contained in each day. Each day, i, of the month has a portion b_i = day's volume / A.

g. For Gauge A, multiply the month's total volume by the portion of each day from Gauge B. i.e. each day's volume (a_i) is:
$a_i = A \times b_i$
Convert back volume to discharge in m^3/s.

A worked example is given in Appendix Table 9.

B.2. METHOD 2: DATA GAPS OF MORE THAN A MONTH AND REFERENCE GAUGE AVAILABLE

a. As for Method 1: Find the total volume for Gauge B.
b. n/a
c. n/a
d. n/a
e. Estimate the total month's volume, *A*, for Gauge A for the missing month using either:
   - The regression relationship between it and Gauge B.
   - The regression relationship between it and rainfall (only SAWS Paarl 1 [0021823 0] was available for this).
   - Or a combination (i.e. an average) of the two estimates.

   The gauge with the best relationship with Gauge A was found by testing various combinations. Where necessary, relationships were found between the two gauges for different periods, in order to ensure that the best relationship and estimate of volume was used for each patch.
f. As for Method 1: For Gauge B, find the portion of total volume contained in each day. Each day, *i*, of the month has a portion b_i = day's volume / *B*.
g. As for Method 1: For Gauge A, multiply the month's total volume by the portion of each day from Gauge B. i.e. each day's volume (a_i) is:

   $a_i = A \times b_i$

Convert back to discharge in m^3/s.

A worked example is given in Appendix Table 10

B.3. METHOD 3: DATA GAPS OF MORE THAN A MONTH AND A REFERENCE GAUGE NOT AVAILABLE

a. n/a
b. n/a
c. n/a
d. n/a
e. The total volume for the missing month of Gauge A was estimated using:
   - The regression relationship between it and rainfall (SAWS Paarl 1 [0021823 0]).
f. Once the total volume was found, in order to apportion the month's volume to each day, a standard "average" pattern was used for wet or dry months. The pattern was determined using the daily flow data for relevant dry or wet season months from Gauge A, from periods where data was available. The "average" shape used (of daily proportions of monthly flow) is shown in Appendix Figure 1 and compared with actual proportions for a wet month for G1H004 and the same month for G1H003.
g. For Gauge A, multiply the volume of each day by the fraction of each day from the standard pattern developed for Gauge A. i.e. each day's volume (a_i) is:

   $a_i = A \times b_i$, where b_i in this case is determined from the standard pattern.

A worked example is given in Appendix Table 11

Appendix Table 9 Method 1: Data gaps less than a month (G1H004); appropriate flow gauge available for patching (G1H020)

	G1h004				G1h020		
Date	m3/s	Volume Mm^3	Patched volume Mm^3	Patched m3/s	m3/s	Volume Mm^3	Portion of month's volume
01/06/1986	2.339	0.202	0.202	2.339	5.5	0.475	07
02/06/1986			2.179	25.224	46.43	4.011	0.062
03/06/1986			2.131	24.660	45.39	3.922	0.060
04/06/1986			0.892	10.322	19	1.641	0.025
05/06/1986			0.759	8.785	16.17	1.397	0.022
06/06/1986	4.083	0.353	0.353	4.083	10.99	0.950	0.015
07/06/1986	2.963	0.256	0.256	2.963	8.388	0.725	0.011
08/06/1986	2.331	0.201	0.201	2.331	6.971	0.602	09
09/06/1986	1.915	0.165	0.165	1.915	6.016	0.520	08
10/06/1986	3.468	0.300	0.300	3.468	7.697	0.665	0.010
11/06/1986	7.055	0.610	0.610	7.055	11.59	11	0.015
12/06/1986	37.703	3.258	3.258	37.703	71.53	6.180	0.095
13/06/1986	21.076	1.821	1.821	21.076	49.69	4.293	0.066
14/06/1986	14.539	1.256	1.256	14.539	26.88	2.322	0.036
15/06/1986	12.194	1.054	1.054	12.194	19.81	1.711	0.026
16/06/1986	11.094	0.959	0.959	11.094	16.92	1.462	0.022
17/06/1986	23.836	2.059	2.059	23.836	35.65	3.080	0.047
18/06/1986	18.754	1.620	1.620	18.754	33.33	2.879	0.044
19/06/1986	14.278	1.234	1.234	14.278	23.98	2.072	0.032
20/06/1986	34.258	2.960	2.960	34.258	65.21	5.634	0.087
21/06/1986	19.959	1.724	1.724	19.959	44.73	3.865	0.059
22/06/1986	14.245	1.231	1.231	14.245	26.12	2.257	0.035
23/06/1986	12.268	1.060	1.060	12.268	20.68	1.787	0.028
24/06/1986	11.146	0.963	0.963	11.146	18.06	1.560	0.024
25/06/1986	10.618	0.917	0.917	10.618	16.44	1.421	0.022
26/06/1986	10.1	0.873	0.873	10.100	15.26	1.318	0.020
27/06/1986	9.885	0.854	0.854	9.885	14.18	1.225	0.019
28/06/1986	9.681	0.836	0.836	9.681	13.29	1.148	0.018
29/06/1986	17.23	1.489	1.489	17.230	33.52	2.896	0.045
30/06/1986	12.515	1.081	1.081	12.515	22.54	1.948	0.030
TOTAL:		**29.336**	**35.2965**			**64.967**	

Days missing in Gauge A (G1H004) are 2, 3, 4, and 5 June 1986.

a. Total volume for June 1986 for Gauge B (G1H020) (last row in Table): **B = 64.967 Mm^3**.

b. Total volume for G1H020, of all days excluding 2 to 5 June 1986: **x = 53.995 Mm^3** (and volume of 2 to 5 June in Gauge B = 10.972 Mm^3).

c. Total volume for Gauge A, of all days other than 2 to 5 June 1986: **y = 29.336 Mm^3**.

d. Ratio R of total volume to "not-missing-days" for Gauge B: **R** = **B/x = 64.967** / **53.995 = 1.2032**.

e. Total month's volume, for Gauge A (A) = volume not-missing-days (y) x ratio R i.e.:
$A = y$ x B/x = 29.336 x **1.2032 = 35.2965 Mm^3.**

Volume missing days (z) for Gauge A = total volume (A) - volume not-missing-days (y). i.e.

$z = A - y$ = 35.2965 - 29.336 = 5.961

f. For Gauge B, the portion of total volume contained in each day, i, of the month: b_i = day's volume / A.
e.g. the volume for 02/06/1986 for Gauge A = 4.011 (see last column of Table).
Thus: **b_i = 4.011 / 64.967 = 0.062**

g. For Gauge A, multiply the month's total volume by the portion of each day from Gauge B.
i.e. each day's volume = a_i = A x b_i.. Thus, for 02/06/1986: a_i = **35.2965** x **0.062 = 2.179 Mm^3**
Convert back to discharge in m^3/s = 2.339 m^3/s.

As a matter of interest, the total volume A estimated by the regression equation for the relationship between monthly volumes at G1H020 and G1H004 (for the period March 1966 to Apr 2007 (the end of G1H004's record): $y = 0.36x + 3.0268$, $R^2 = 0.7609$) was **26.4148** Mm^3, as compared to the 35.2965 Mm^3 estimated in the above approach.

Appendix Table 10 Method 2: Data gap more than a month (G1H004); reference gauge available (G1H003)

	G1h004				G1h003		
Date	m3/s	Volume Mm^3	Patched volume Mm^3	Patched m3/s	m3/s	Volume Mm^3	Portion of month's volume
01/06/1954			5.6218	65.0667	8.153	0.7044	0.0555
02/06/1954			5.3508	61.9303	7.76	0.6705	0.0528
03/06/1954			3.9559	45.7853	5.737	0.4957	0.0390
04/06/1954			3.1222	36.1367	4.528	0.3912	0.0308
05/06/1954			2.8871	33.4152	4.187	0.3618	0.0285
06/06/1954			2.8871	33.4152	4.187	0.3618	0.0285
07/06/1954			2.6678	30.8774	3.869	0.3343	0.0263
08/06/1954			2.3499	27.1983	3.408	0.2945	0.0232
09/06/1954			2.2513	26.0570	3.265	0.2821	0.0222
10/06/1954			2.1582	24.9796	3.13	0.2704	0.0213
11/06/1954			2.0658	23.9102	2.996	0.2589	0.0204
12/06/1954			1.9548	22.6253	2.835	0.2449	0.0193
13/06/1954			1.8445	21.3484	2.675	0.2311	0.0182
14/06/1954			1.8107	20.9573	2.626	0.2269	0.0179
15/06/1954			1.9217	22.2422	2.787	0.2408	0.0190
16/06/1954			2.3816	27.5654	3.454	0.2984	0.0235
17/06/1954			2.3816	27.5654	3.454	0.2984	0.0235
18/06/1954			1.9948	23.0882	2.893	0.2500	0.0197
19/06/1954			1.9010	2228	2.757	0.2382	0.0188
20/06/1954			1.8280	21.1569	2.651	0.2290	0.0180
21/06/1954			1.7452	20.1992	2.531	0.2187	0.0172
22/06/1954			1.6811	19.4570	2.438	0.2106	0.0166
23/06/1954			1.6494	19.0899	2.392	0.2067	0.0163
24/06/1954			4.0414	46.7749	5.861	0.5064	0.0399
25/06/1954			6.3168	73.1113	9.161	0.7915	0.0623
26/06/1954			11.9124	137.8748	17.276	1.4926	0.1175
27/06/1954			6.9519	80.4615	10.082	0.8711	0.0686
28/06/1954			5.7700	66.7826	8.368	0.7230	0.0569
29/06/1954			4.2282	48.9377	6.132	0.5298	0.0417
30/06/1954			3.7490	43.3911	5.437	0.4698	0.0370
TOTAL:			**101.3821**			**12.703**	

Days missing in Gauge A (G1H004) : all of June 1954 missing

a. Month's volume for Gauge B (G1H003) = 12.703 Mm^3
b. n/a
c. n/a
d. n/a
e. The total volume for the missing month of G1H004 was estimated using the average of the estimations of the volume from:
   - The relationship between monthly volumes at G1H004 and G1H003 during this period ($y = 13.118\, x^{0.9579}$; $R^2 = 0.8015$) $y = 13.118 \times 12.703^{0.9579} = 149.731$,
   - The regression relationship between monthly volume at G1H004 and monthly rainfall at SAWS Paarl 1 [0021823 0] ($y = 0.5837\, x + 0.5$; $R^2 = 0.7524$). $y = 0.5837 \times 90 + 0.5 = 53.033$,
   - giving an average volume for the month for G1H004 = 101.382 Mm^3
f. Find each day's portion of the month's volume for Gauge B:
   e.g. For G1H003, the portion of the month's volume for 01/06/1954:
   = 0.704 / 12.703 = 0.0555.
g. Thus, the volume for 01/06/1954 for G1H004 was 0.0555 x 101.382 = 5.621 Mm^3.
   Convert back to discharge in m3/s = 65.0667 m^3/s.

Note: Where a patch generated values above the rating, the data were deleted, and Method 1 was applied.

Appendix Table 11 Method 3: Data gap more than a month (G1H004); reference gauge was NOT available

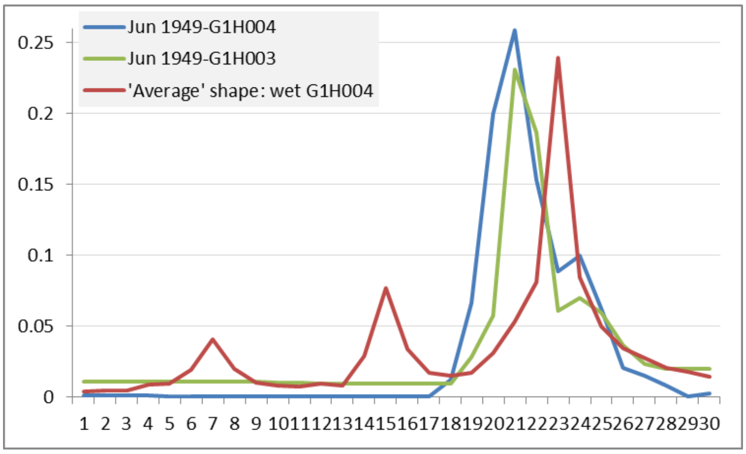
	G1h004				"Standard" wet month pattern		
Date	m3/s	Volume Mm^3	Patched volume Mm^3	Patched m3/s			Portion of month's volume
01/06/1954			0.1673	1.9369			039
02/06/1954			0.1988	2.3013			046
03/06/1954			0.2097	2.4275			049
04/06/1954			0.3803	4.4013			088
05/06/1954			0.4138	4.7890			096
06/06/1954			0.8202	9.4930			0.0190
07/06/1954			1.7486	20.2389			0.0405
08/06/1954			0.8692	10.0607			0.0201
09/06/1954			0.4415	5.1102			0.0102
10/06/1954			0.3578	4.1415			083
11/06/1954			0.3147	3.6425			073
12/06/1954			0.4269	4.9411			099
13/06/1954			0.3572	4.1341			083
14/06/1954			1.2464	14.4254			0.0289
15/06/1954			3.3174	38.3963			0.0768
16/06/1954			1.4648	16.9536			0.0339
17/06/1954			0.7541	8.7281			0.0175
18/06/1954			0.6398	7.4046			0.0148
19/06/1954			0.7548	8.7358			0.0175
20/06/1954			1.3545	15.6774			0.0314
21/06/1954			2.2912	26.5185			0.0531
22/06/1954			3.5095	40.6193			0.0813
23/06/1954			10.3431	119.7118			0.2396
24/06/1954			3.6561	42.3156			0.0847
25/06/1954			2.1564	24.9582			0.0500
26/06/1954			1.4813	17.1451			0.0343
27/06/1954			1.1968	13.8516			0.0277
28/06/1954			0.8942	10.3498			0.0207
29/06/1954			0.7860	9.0974			0.0182
30/06/1954			0.6159	7.1286			0.0143
TOTAL:			**43.1685**				

Days missing in Gauge A (G1H004) : all of June 1952 missing

a. n/a
b. n/a
c. n/a
d. n/a
e. The total volume for the missing month of G1H004 was estimated using:
   - The regression relationship between monthly volume at G1H004 and monthly rainfall at SAWS Paarl 1 [0021823 0] (y = 0.5837 x + 0.5; R^2 = 0.7524).

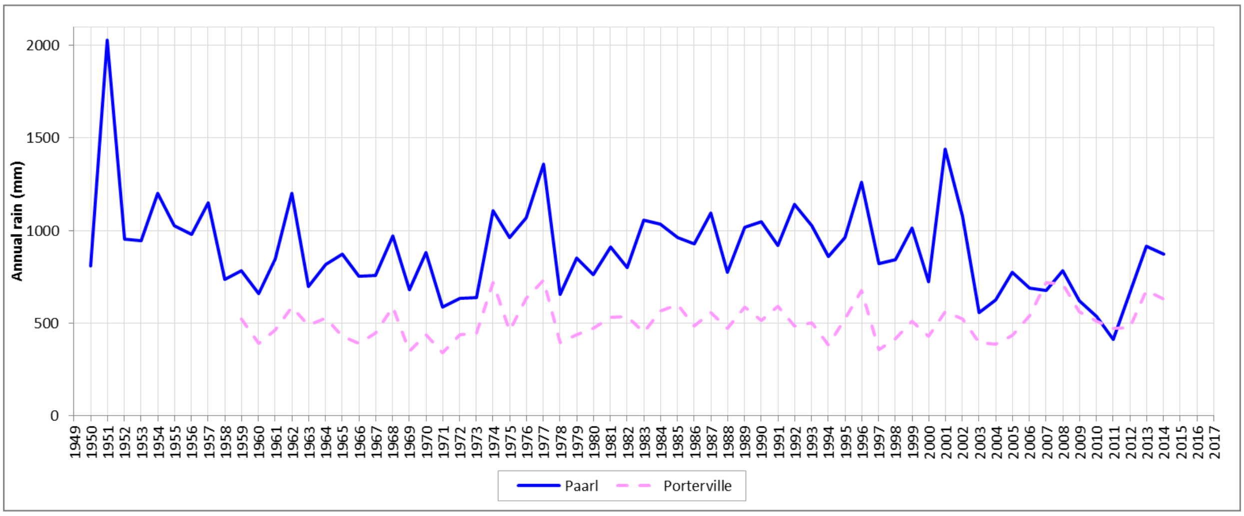
     y = 0.5837 x *73.1* + 0.5 = 43.1685,
f. A "standard" wet month pattern, for G1H004, for the proportion of total month's volume for each was estimated for G1H004 (last column in table):
   e.g. For G1H004, the portion of the month's volume for the first day = 039.
g. Thus, the volume for 01/06/1952 for G1H004 was 0. 0039 x 43.1685 = 0.1673 Mm^3.
   Convert back to discharge in m3/s = 1.936 m^3/s.

Note: Where a patch generated values above the rating, the data were deleted, and Method 1 was applied.



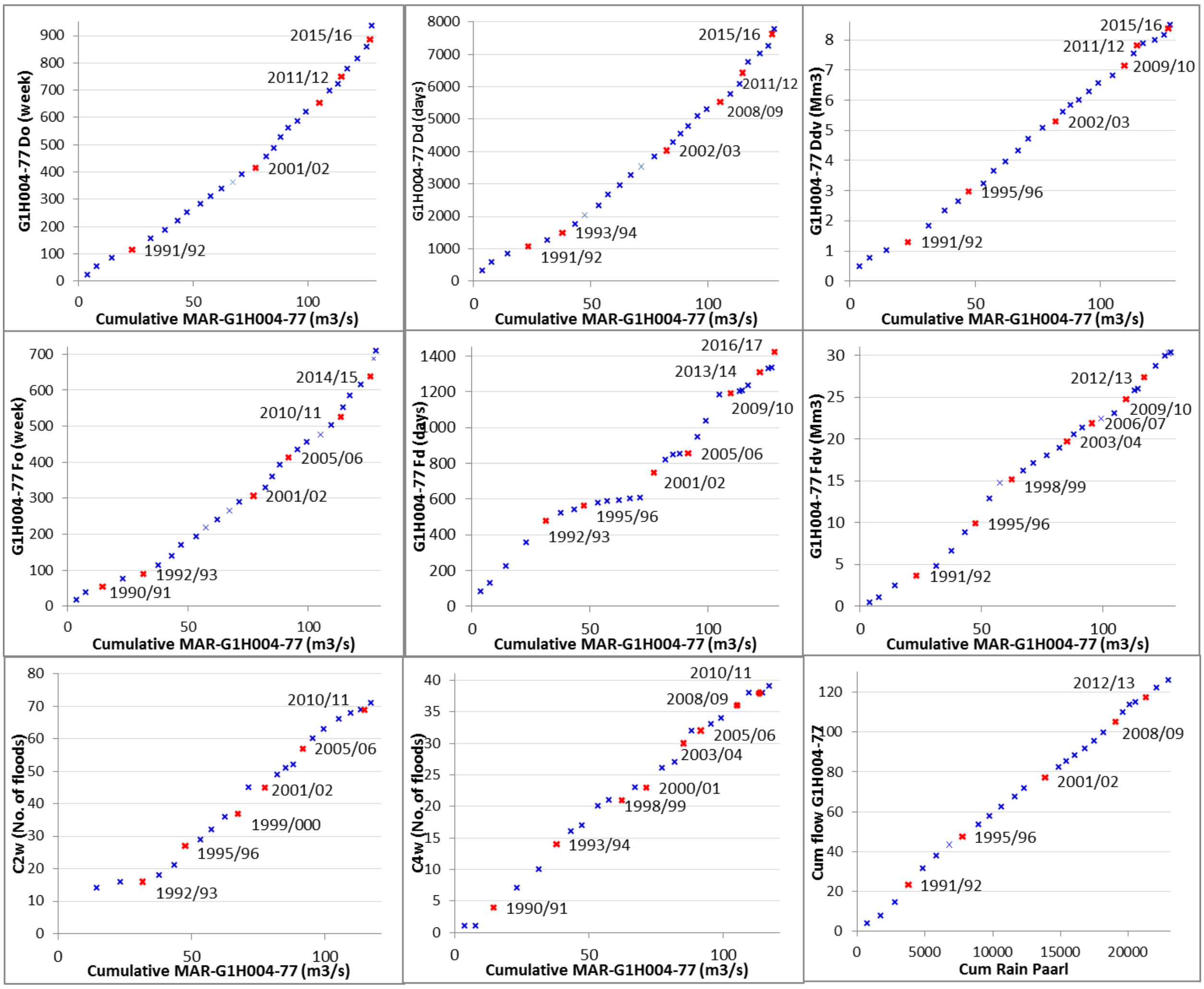
Appendix Figure 1 A comparison of actual daily proportions of monthly flow for June 1949 for G1H004, G1H003 and the "average" shape for G1H004 for the wet season.

B.4. PAARL AND PORTEVILLE ANNUAL RAINFALL COMPARISON

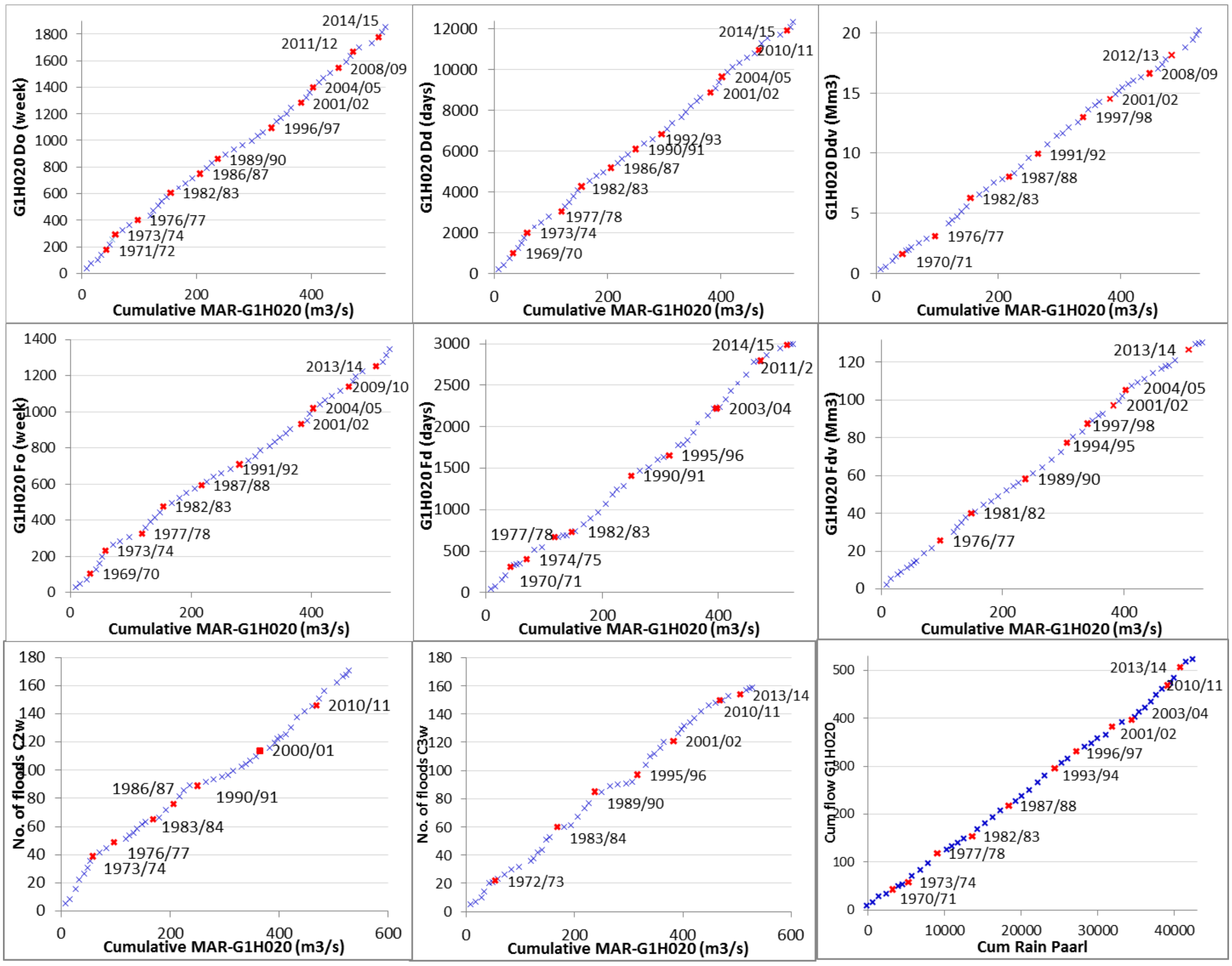


Appendix Figure 2 Annual rainfall volume for Paarl and Porterville gauges

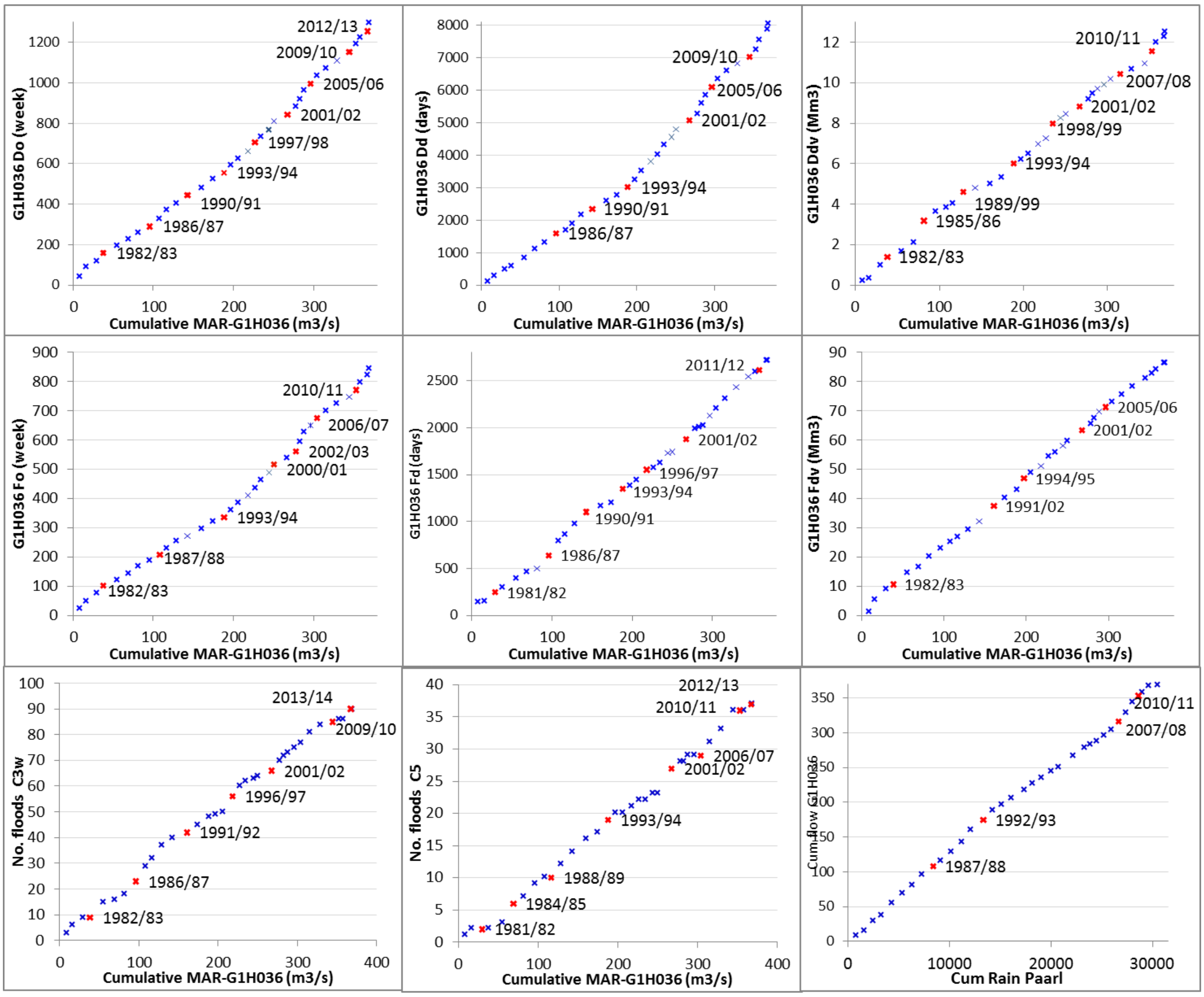
B.5. DOUBLE MASS PLOTS WITH MARKED INFLECTION POINTS



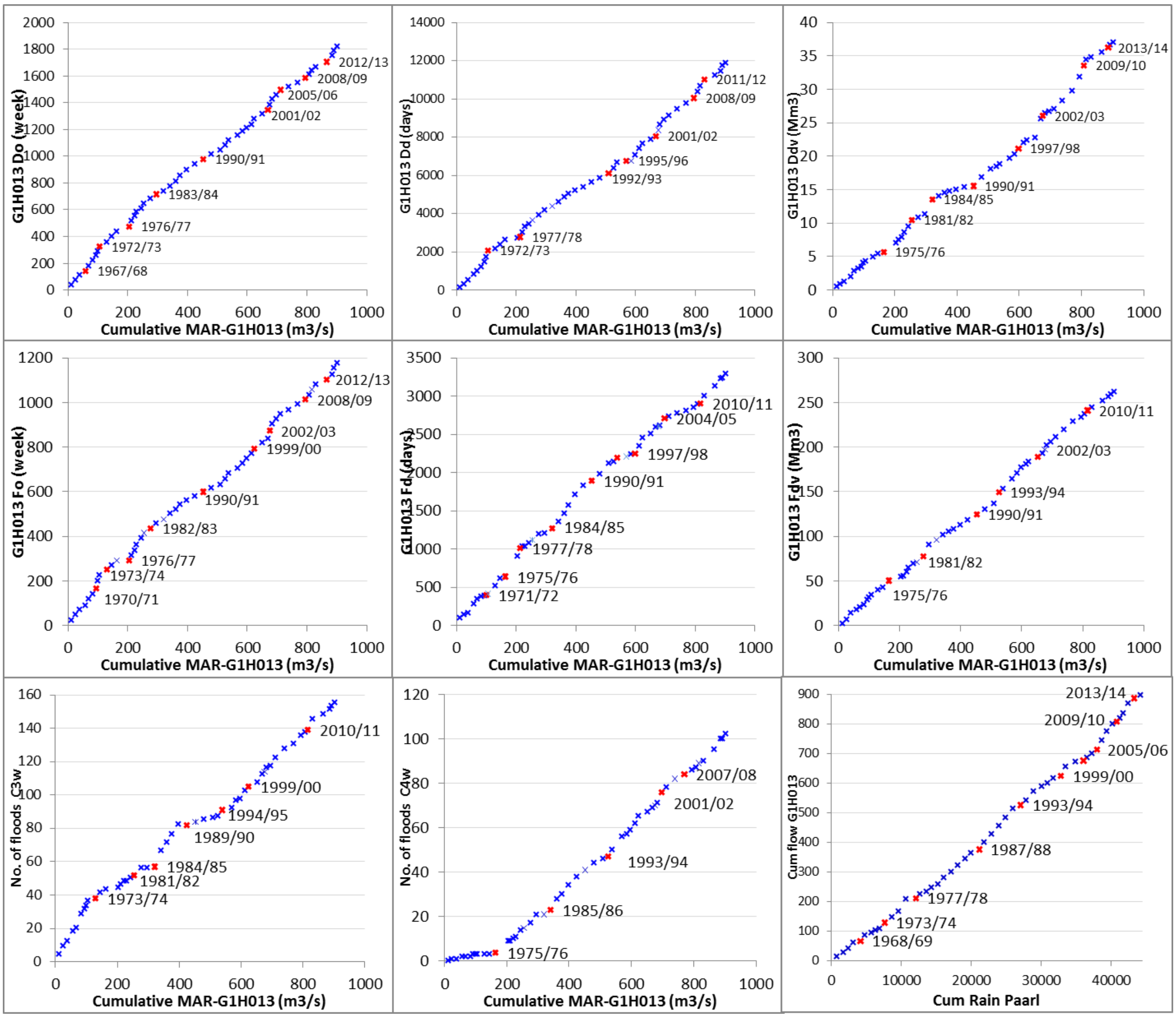
Appendix Figure 3 Double mass plots of G1H004 and G1H077 combined MAR against dry season onset (Do), dry season duration (Dd), dry season volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season volume (Fdv), the number of Class 2 (C2w), Class 4 (C4w) floods and cumulative rainfall



Appendix Figure 4 Double mass plots of G1H020 MAR against dry season onset (Do), dry season duration (Dd), dry season volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season volume (Fdv), the number of Class 2 (C2w), Class 3 (C3w) floods and cumulative rainfall



Appendix Figure 5 **Double mass plots of G1H036 MAR against dry season onset (Do), dry season duration (Dd), dry season volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season volume (Fdv), the number of Class 3 (C3w), Class 5 (C5) floods and cumulative rainfall**



Appendix Figure 6 Double mass plots of G1H013 MAR against dry season onset (Do), dry season duration (Dd), dry season volume (Ddv), wet season onset (Fo), wet season duration (Fd), wet season volume (Fdv), the number of Class 3 (C3w), Class 4 (C4w) floods and cumulative rainfall

Appendix C. CHAPTER 7

C.1. COLLATION AND ASSESSMENT OF DATA ON LAND-USE: METHOD STATEMENT

Table 10.1 Collation and assessment of data on land-use

Method	Data collection/collation	Equipment	Data processing	Data format	Data analysis
• Collation and assessment of data on basin land-use	• Work with staff at Department of Surveys and Mapping to identify historical topographic maps that cover the entire river basin • For these periods, obtain the historical topographic maps in electronic format from the Department (there is no charge if you supply the DVD/flash drives) • Copy the data files into a folder maps and launch the *.TIF images in GIS	• Flash drive (~64 GB); computer with MS Excel; a GIS Package (QGIS freeware was used in the Berg River Basin)	• Create a QGIS project file for the river basin and load the following covers: a. DWS quaternary shapefile b. 1:500 000 rivers c. Dams and water bodies d. DWS gauging stations e. Towns and roads • Identify suitable categories of land-use • Create a shapefile for each category land-use (Section 3.3.1 and Table 3.3) and capture: • Polygons for areas; • Points for single features; • Use the built in scripts (these vary between GIS packages) to calculate areas of polygons and count points • Generate maps of land-use and summary tables of area and or point counts	• GIS shapefiles and MS Excel spreadsheets comprising areas of polygon features (km^2), counts of point data (nominal)	• Test for differences in areas and counts between periods • Create summary bar graphs of pertinent changes • Create summary tables of results shown to be different

C.2. COLLATION AND ASSESSMENT OF HYDROLOGICAL DATA: METHOD STATEMENT

Table 10.2 Collation and assessment of hydrological data

Method	Data collection/collation	Equipment	Data processing	Data format	Data analysis
Collation and assessment of hydrological data	• Go to www.dwaf.gov.za to find gauging weirs in the river: • Access daily hydrological discharge data for identified gauges by selecting the gauge number and daily average flow m3/s • Download the entire record • Provide the coordinates for the location of gauging weirs identified above to the South African Weather Service and they will provide rainfall data from nearby rain gauges. (A fee is payable for access to these data)	• Computer with MS Excel; flow analysis software (DRIFT was used in the Berg River Basin)	• Patch the hydrological data: • For gaps longer than one month, patch using data from a nearby reference gauge based on relative MAR • For gaps longer than one month, patch using regression coefficients • See Magoba (2018) for a detailed description and worked examples of patching • Calculate time-series of ecologically-relevant summary statistics • Follow steps described in the user manual for flow analysis software (e.g., for DRIFT this is available on www.DRIFT-EFlows.co.za)	• DRIFT project file; summary statistics for flow indicators, MS Excel spreadsheets, discharge (m^3/s) time series, rainfall (Mm^3) time series,	• Export statistics for ecologically meaningful flow indicators from DRIFT • Plot double-mass plots of MAR against rainfall and selected DRIFT flow indicators • Calculate T-tests to determine when changes in flow, rainfall and DRIFT indicators took place • Compare time periods with the timing of changes in land-use or water resource developments

C.3. COLLATION AND ASSESSMENT OF CHANNEL CHANGE AND RIPARIAN VEGETATION DATA: METHOD STATEMENT

Table 10.3 Collation and assessment of channel change and riparian vegetation data

Method	Data collection/collation	Equipment	Data processing	Data format	Data analysis
Collation and assessment of channel change and riparian vegetation data	• Go to Department of Surveys and Mapping with blank DVDs and ask to see the flight plans for aerial images across the river basin • Select images that correspond to the periods selected (Table 7.4) and the location of sites • Copy the data files into a folder *aerial images* • Download **Image Composite Editor** from www.microsoft.com/en-us/download/ • Open the images from each site and select *new panorama* to load and stitch the images together • Open Google Earth® and capture recent images for the same sites, saved as *.JPEG • Load the stitched historical and Google Earth images into MS Power Point per time period and site	• Flash drive (~64 GB); computer with MS Excel	• Select a rigid and permanent feature on the image, such as a road, or a bridge, and mark it out on each image • Use the dimensions and orientation of this feature to size the images to the same scale and direction as one another • Capture a line along the thalweg and polygons of the alluvial bars, riparian area and floodplain from each historical image • Select one of the Google Earth® images and open the slide show alongside a view of the site in Google Earth® • Capture the thalweg line and polygons and record the distance in m and the area in m^2 • Measure a straight line along the distance of the thalweg and record the distance in metres	• MS Power Point files, MS Excel spreadsheets, lengths of channel (m^2), area of polygon features (m^2)	• Calculate the relative length of lines and the proportional area of polygons for each site in MS Excel • Use the ratio of proportions between years per site to calculate ACTUAL lengths and areas using those measured for one of the recent images in Google Earth® • Calculate sinuosity; divide the thalweg length by the straight line length • Test for differences in areas and lengths between time periods per site

C.4. COLLATION AND ASSESSMENT OF DATA FROM MACROINVERTEBRATE SURVEYS: METHOD STATEMENT

Table 10.4 Collation and assessment of data from the macroinvertebrate surveys

Method	Data collection/collation	Equipment	Data processing	Data format	Data analysis
Collation and assessment of data from macro-invertebrate surveys	• Source and collate community data from historical surveys, and organise the data into sample columns and row list of taxa • For current data, the samples are collected using same procedures as SASS – so if SASS assessments are being done for REMP it is only necessary to collect samples for processing • To collect the samples, drain water from sample through sieve, and empty tray contents into a sample jar, preserve with 96% ethanol diluted with river water to a 70% solution • Mark sample clearly and store sample out of the sun, in a dark cupboard, until processed	• Data collection: Hand-held 1-mm mesh net; sampling tray, forceps, SASS5 identification guide, 1-mm mesh sieve, 96% chemical grade ethanol, 500 ml sample jars, plastic vegetable bags, masking tape, alcohol proof marking pen, pencil, white paper • Data processing: Dissecting microscope, tray, 5-ml sample storage tubes • Data analysis: computer with MS Excel, statistical analysis package (PRIMER Microsoft Excel was used in the Berg River Basin)	• New samples: a. Sieve the sample to remove debris and stones b. Float organisms in distilled water and separate into family groups c. Enter data into spreadsheets in format used for historic data • All data: a. Cross-check family names to ensure that any changes have been captured and families are correctly named b. Format data into PRIMER project files according to instructions in PRIMER manual	• MS Excel spreadsheets; counts of families, SASS scores, PRIMER project files, graphs and tables of community abundance similarity and dissimilarity	• Calculate total SASS5 score and average score per taxon (ASPT) per biotope, and assign condition: • Use Dallas (2007) to calculate a condition score • Load the spreadsheet of family abundance per site per year into PRIMER and enter factor codes. For each site: • Run an ANOSIM test for differences between years • Create CLUSTER graphs of similarity between years • Run SIMPER analysis to determine the organisms responsible for similarity and dissimilarity between years.