Spontaneous succession of riparian vegetation and aquatic macroinvertebrates along the Silvermine River, South Africa, after fire and clearing of exotic plant species

by

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"Human kind should act in such a manner that is a part of Nature rather than apart from nature"

Mahatma Gandhi

ABSTRACT

Spontaneous succession, as a method to restore degraded riverine habitats, was assessed for three different components of the Silvermine River ecosystem over one year. These were the state of the physical habitats (biotopes) available to aquatic organisms, changes in the community composition of riparian vegetation, and changes to aquatic macroinvertebrate assemblages present in the river. Data were collected during two successive summer-sampling periods (2001 and 2002) at three study sites along the river. Site 1 was situated in the mountain stream zone, while Sites 2 and 3 were situated in the foothills. Changes in the riverine ecosystem and its associated habitats were compared to reference condition data for each of the three components from each study site. There were no large changes to the macro-channel banks of the river over the one year study, other than where a meander cut-off occurred at Site 2. Changes over the year in the proportions of flow at the three sites revealed a lack of faster-flow types, which would normally characterise mountain stream and foothill reaches, at all three study sites. Measurements of the proportions of substratum types at the three sites revealed there was an unusually high proportion of fine substratum types present. In the foothills, this was attributed to erosion of the unstable bank at Site 2, which continued to deposit an excess of fine sediment that was transported downstream. In the vegetation study, most species that came to dominate the mountain stream (Site 1) and the foothill (Sites 2 and 3) riparian communities emerged from the seed bank within the first two years after the fire. Thus, using the presence or absence of characteristic riparian vegetation species it is possible to determine whether there is a need to augment the recovery process after two years. The aquatic macroinvertebrate communities of the mountain stream differed clearly from those in the foothill. The differences were attributed to a combination of the channel type and the longitudinal position of the sites along the river, both of which dictated the physical habitats available. There were clear differences in the potential for recovery at each of the three study sites. It was shown that non-intervention will not support successful recovery in the short to medium term for any of the measured three components of this river. A blanket policy of non-intervention cannot address site-specific differences, be they natural or artificial, which pose different challenges to restoration. There were zonal differences in the availability of substrata and flow types. There were also other differences that were attributed to water abstraction and the presence of different woody exotic trees. Recovery of the mountain stream was being hampered by the presence of the reservoir and the history of water abstraction while recovery of the foothill was being retarded by erosion from the massive sediment deposit at Site 2. Plans to restore the river, that took into account these inter-site differences, were proposed.

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1. INTRODUCTION

Species extinctions and habitat destruction continue due to humanity's failure to acknowledge its dependence upon global ecosystems (Perrow and Davy 2002). The recent emergence of *restoration ecology* as a science, over the last two decades, has largely been driven by a need to collate studies and experiments that have attempted to restore degraded ecosystems, or parts thereof. Although Aldo Leopold's pioneering reconstruction of tall-grass prairie at the University of Wisconsin Arboretum is considered by many to be the first international ecologically based restoration in North America (Higgs 1997), restorations have been practiced in some form or other for centuries (Palmer *et al.* 2006), even before the advent of the industrial revolution. The term restoration is often defined as 'the re-establishment of *pre-disturbance* ecological functions and related physical, chemical, and biological characteristics (Palmer *et al.* 2006). This requires a holistic approach that cannot be achieved through the manipulation of isolated ecosystem elements (Council 1992). The increase in scientific investigation into ecological restoration is reflected in the establishment of the Society for Ecological Restoration (SER) in 1987 and their subsequent publishing of the journal *Restoration Ecology* from 1993 onwards. As an inherently multi-disciplinary science, ecological restoration encompasses a broader scientific community than the disciplines closer to its ecological core: there are economic, cultural, political, social and physical environmental influences (Perrow and Davy 2002). South African scientists are developing skills in this new arena and are following the growing global community of restoration practitioners in a drive to develop principles from the experimental restoration practices currently underway. Restoration science should be practised and applied in all terrestrial and aquatic ecosystems globally. In this thesis, the focus is on the restoration of some aspects of the flora and fauna of lotic ecosystems: specifically, riparian vegetation and aquatic macroinvertebrates.

1.1 Restoration paradigms

Although conservation authorities and organizations exist in most developed countries and at a global level, there remains insufficient protection for undisturbed ecosystems against modern society's destructive practices that eliminate species and habitats. The failures, in part, are not necessarily the fault of the people or the organisations involved but may have more to do with the way people view ecosystems.

Initially restoration projects probably had the general aim of returning an ecosystem back to a close approximation of its condition prior to disturbance. As restoration projects evolved, practitioners became focussed upon defining targets for their restoration activities. For example, during the latter half of the 20th century debate was initiated on what constitutes restoration and how it might differ from rehabilitation. Four terms were largely considered: restoration, rehabilitation, and remediation/enhancement (see section 2.6). These terms were repeatedly defined according to the level that an ecosystem was restored back towards its original state. There were two schools of thought: those following the traditional Balance of Nature paradigm and those aligned with modern Flux of Nature paradigm (Rogers and Bestbier 1997).

The traditional Balance of Nature paradigm was the basis for 20th century conservation practices; it followed the view that ecosystems were static and in equilibrium if not subject to disturbance. Conservation programmes focused on maintaining stable landscapes by managing desired species at chosen population levels. This was achieved by excluding anthropogenic influences and controlling populations of species in enclosed areas. Proponents of the Balance of Nature view believe that humans exist outside of what is considered to be a natural ecosystem (Jackson *et al.* 1995) unless the humans are indigenous peoples or hunter-gatherers. In this context, human influences are undesirable, and a natural ecosystem is considered to be one that has not been impacted by modern society in any way. The protection of desired ecosystems from all natural and anthropogenic influences external to the conserved ecosystem resulted in a general reduction in natural variation in the managed system, and a loss of ecosystem resilience that maintained ecosystem functioning at a chosen level (Rogers and Bestbier 1997).

The modern Flux of Nature paradigm recognises that ecological systems are rarely in equilibrium, but are dynamic and heterogeneous in nature. Since ecosystems are continually changing, the biological constituents present today in an ecosystem will not necessarily be the same as those present in the future (Rogers and Bestbier 1997). Ecosystems are thus seen as being temporally unique, sculpted by isolated climatic events and biological invasions (Jackson *et al.* 1995). This more modern view impacts on the way natural ecosystems are managed, as the focus shifts towards an ecosystem-level perspective where temporal and spatial heterogeneity is emphasised across all scales from genes to landscapes (Rogers and Bestbier 1997). The Flux of Nature paradigm recognises that anthropogenic influences on natural landscapes are unavoidable and in fact necessary for sustainable management. Manipulations of ecosystem components are forms of landscape engineering, regardless of whether or not they are designed to achieve some natural state. This engineering may be passive, where natural forces are allowed to operate, or active. All management is located somewhere on the continuum from active engineering to pure custodianship of ecosystems. Man is seen as an integral part of ecosystems, and the landscape is a manifestation of value judgements that reflect a mix of anthropogenic and natural forces. This contrasts with the traditional Balance of Nature view, where any human influence or impact on a natural ecosystem is seen as a disturbance with a negative influence (Jackson *et al.* 1995).

The Flux of Nature paradigm must be interpreted with caution as it appears to license anthropogenic manipulation of ecosystem processes. Although modern thinking considers human intervention necessary for conservation and management of natural landscapes, anthropogenic influences are generally still seen as being detrimental to natural ecosystems. Thus, many custodians of global ecosystems probably hold a view somewhere between the two paradigms. Dynamism and heterogeneity are seen as essential ecosystem attributes yet unplanned human influences are still seen to impact on natural landscapes negatively.

1.2 Rationale

In South Africa, government-owned wilderness areas are usually managed by the South African National Parks service (SANP). In the Western Cape, a province of South Africa, there is an additional wilderness management authority, CapeNature, which was previously known as Cape Nature Conservation (CNC). The two largest perennial rivers on the Cape Peninsula, near Cape Town, are the Disa and the Silvermine Rivers. The lower part of the Disa River catchment is a developed residential suburb known as Hout Bay. The Silvermine River catchment is largely undeveloped and, up until 1998, the larger part was owned by the Cape Town City Council and managed by CapeNature. It was known as the Silvermine Nature Reserve and the upper reaches were afforested with exotic pine trees (*Pinus radiata*) and used extensively by residents of Cape Town for recreational activities. The middle catchment was farmed with agricultural crops or livestock. The valley slopes of the middle reaches are still largely covered with exotic kikuyu grass (*Pennisetum clandestinum*) on man-made terraces, and dense stands of invasive exotic *Acacia* spp. persisted after the cessation of farming practices in 1968 (Table Mountain National Park *pers. comm.*). The only developed areas within the boundaries of the catchment are the residential suburb of Clovelly and the Clovelly Country Club, both on the coastal floodplain and outside the Silvermine Nature Reserve.

All undeveloped areas of the Silvermine River catchment were incorporated into the Cape Peninsula National Park (CPNP) in May 1998, and the old Silvermine Farm homestead was established as the headquarters for the CPNP personnel. In time, other undeveloped sections of the Cape Peninsula around Cape Town were merged together and were added to the CPNP. In 2005 these areas, along with the Silvermine River catchment but excluding its urbanised coastal floodplain, were all collectively renamed the Table Mountain National Park (TMNP) that is now managed by SANP.

In early January 2000, wild fires burnt large areas of the Cape Peninsula including most of the Silvermine River catchment. All undergrowth and sub-canopy floral communities were incinerated; few large adult trees were left standing. The incineration of large portions of the predominantly alien vegetation in the Silvermine catchment offered the SANP an opportunity to begin an alien eradication programme. The clearing was done by Working for Water (van Wilgen *et al.* 1998) personnel and managed by SANP. As such, in line with Working for Water policies at the time, no re-vegetation strategies were employed to enhance re-vegetation of the area.

This exclusion of anthropogenic influence in the process of recovery followed a Balance of Nature paradigm, which was the view held by the conservation authorities in South Africa at the time. Thus planned interventions, other than the alien clearing, that would disrupt the natural processes of recovery were purposefully avoided. This process of un-aided ecosystem recovery has been termed spontaneous succession (Prach *et al.* 2001). The assumption behind choosing to adopt recovery via spontaneous succession rather than intervening is that natural processes of recovery are sufficient to restore previously disturbed habitats back towards their undisturbed state. Since there were few other external negative influences on the state of the Silvermine River ecosystem, this situation presented an opportunity to study whether spontaneous succession may be employed as a restoration measure successfully.

1.3 Objective

The overall objective of this research was to record spontaneous succession along the Silvermine River after the January 2000 fires and during the catchment disturbances associated with the clearing of the exotic plant species. The hypothesis was that unassisted pathways of recovery alone are sufficient to restore the Silvermine River and its associated habitats to some resemblance of its natural state, as assessed through comparison with neighbouring undisturbed rivers. This hypothesis was applied to three components of the river ecosystem: the state of physical aquatic habitats, riparian vegetation community structure and aquatic macroinvertebrate assemblage composition.

The aims were to record:

1. physical characteristics of the macro- and active channel, and to track changes in these and aquatic habitat from one summer period to the next, due to winter floods and instability in the riparian zone and gradual re-growth of vegetation;

2. the nature and re-growth of the riparian vegetation through one year of seasonal changes and alien clearance;
3. the nature of aquatic invertebrate faunal assemblages, a year apart.

Data were collected at three study sites along the river. As each site was situated in a different part of the catchment, the natural processes affecting the biota, and the disturbances associated with the presence and clearing of different exotic species, were different.

1.4 Thesis layout

This thesis contains eight chapters. Chapter two summarises literature relevant to (river) restoration ecology. Chapter three outlines the study area and other features of interest at the study sites. Chapter four presents the data gathering and analysis methods while chapters five through seven provide the results and discussions in three separate topics: physical site characteristics, riparian vegetation and aquatic macroinvertebrates. Finally, a concluding chapter summarises the main findings and highlights pertinent management implications and recommendations.

2. LITERATURE REVIEW

The thesis investigates whether natural processes of recovery alone may restore a lotic ecosystem following disturbance. The river under investigation is situated in the Western Cape. Literature pertaining to community ecology thus relates to rivers in this province of South Africa. Data were collected under three categories: aspects of physical riverine habitat, riparian vegetation and aquatic macroinvertebrate communities. This chapter reviews topics in the three data chapters as well as some aspects of restoration ecology, particularly the philosophy relating to planned interventions.

The first section introduces some basic concepts of river ecology. This is followed by a summary of the hydrological and gemorphological influences on riverine habitat and methods for their measurement. Western Cape riparian vegetation is discussed next, along with the methods used to sample vegetation communities. Aquatic macroinvertebrates are then introduced and their use as indicators of habitat quality is discussed. This is followed by an introduction to numerical methods of data classification. Sections on concepts of restoration and the growth of restoration ecology in South Africa are followed by a discussion on recovery and the use of reference sites in assessing recovery.

2.1 River ecology

Rivers are complex, dynamic, heterogeneous and multidimensional. Natural rivers manifest a diversity of landscape elements (Ward *et al*. 2002). Their ecological nature and functioning are influenced primarily by river size, position in the catchment's drainage network, the hydrologic regime and local geology and geomorphology (Naiman and Decamps 1997). The nature of riverine ecosystems changes along the river's course. Rivers may be small in headwater reaches and completely embedded in forest. They may be wider along middle reaches, forming distinct bands within the landscape that are determined by long-term (>50 yrs) channel dynamics and annual discharge. Also they may contain well developed, physically complex floodplains that experience seasonal flooding and lateral channel migration with diverse vegetation communities and moist soils. Studies of rivers thus recognise three main perspectives:

- longitudinal - from the river source to its mouth;
- vertical - between the bank, river bed and fluvial aquifers;
- lateral across the floodplain.

Limnology (freshwater ecology) considers biophysical factors affecting the distribution and abundance of riverine species, and how the interactions between such species at the ecosystem level regulate nutrient and energy cycles (Benda *et al.* 2002). Key concepts in this field, after Lorenz *et al.* (1997) include:

- zonation, stream hydraulics, river continuum, nutrient spiralling, serial discontinuity, patch dynamics and habitat template. These all deal with the longitudinal distribution of hydrological and geomorphological characteristics that govern river function and structure;
- flood pulse and riverine productivity, which emphasize lateral interactions between the river, riparian zone and floodplain; and
- catchment hierarchy and the telescoping ecosystem model, which integrate longitudinal, lateral and vertical dimensions of catchments.

These theories account for spatial variation adequately but do not provide a rigorous framework for the analysis of restoration options (Richards *et al.* 2002). The 'River Continuum Concept' (Vannote *et al.* 1980) predicted downstream ecological change as a function of changes in channel geometry, riparian vegetation and organic matter. Maximum biodiversity was assumed to occur in the middle river reaches. There was no consideration of temporal changes or lateral and vertical dimensions (Benda *et al.* 2002). Patch dynamics theory described the physical river as a discontinuum, or mosaic, of patches formed by the combination of different environmental conditions and processes of disturbance or succession through time (Poole 2002). The River Continuum Concept and Patch Dynamics are particularly relevant in studies on aquatic macroinvertebrates as they describe changes in the availability of organic matter and physical habitat patches. Swanson *et al.* (1992) separated geomorphic patch forming processes, such as sediment movement, from non-geomorphic processes, such as topographic position. The flood pulse concept (Junk *et al.* 1989) related ecological response to seasonal hydrological change by emphasizing lateral connectivity and the role of seasonal variation in structuring recruitment of obligate riparian species (Richards *et al.* 2002). The telescoping ecosystem model (Fisher *et al.* 1998) viewed rivers as a series of nested subsystems. Recovery from disturbance was therefore seen as a function of the interaction between subsystems. The intermediate disturbance hypothesis (Ward and Stanford 1983) described a trade off between the alternate monopoly of stable habitats held by competitively dominant species, and that of fugitive species tolerant of high instability levels (Richards *et al.* 2002). The flood-pulse concept and intermediate disturbance hypothesis are inter-related through patch dynamics and together are particularly pertinent for riparian systems as they deal with channel dynamics that create a diversity of habitat patch sizes.

These concepts help formulate a useful framework in which to consider data organisation, but do not themselves form a practical management tool based on reliable quantitative data (Harris 1988). Many of the ideas in the concepts listed above have been merged under the umbrella of landscape ecology, which is a sub-discipline of ecology and geography. Landscape ecologists study how the physical environment influences ecological processes. The focus of this discipline is how spatial structure influences the distribution of organisms and their abundance in the environment at the landscape level (www.wikipedia.com 2008). Since restoration ecology deals primarily with habitat re-creation and the re-establishment of biota, the discipline of landscape ecology is suitable for the testing of restoration hypotheses.

The next three sections introduce background information pertaining to the three main data chapters.

2.2 Physical river habitat

The *hydrological regime* is the principle driver that works on *geomorphic features* to produce a dynamic and constantly changing physical template (Lorenz *et al.* 1997). These two principal components of physical river habitats are considered below.

Hydrology

Surface water hydrologists study the hydrological cycle and how hydrological processes influence water quality and quantity (Benda *et al.* 2002). HBN Hynes, the father of contemporary river ecology, emphasised the need to understand hydrological controls on biota (Stanford 1998). He noted that stream power generated by discharge interacted with the landscape by eroding bedrock and terrace soils. In so doing, alluvium was redistributed among riverine habitats thereby maintaining landscape diversity within the rivers. A natural disturbance regime controls landscape evolution by maintaining the erosion, transport and subsequent deposition of sediment. A river's flooding history determines channel morphology through the processes of cut and fill avulsion. This produces a continuum of instream morphologies, such as pools, riffles, runs, gravel bars, avulsion channels, islands, debris dams and lateral floodplain terraces (Stanford *et al.* 1996). Local research on integrated management of river flows has reflected a new understanding of the significance of a river's flow regime in maintaining the nature and functioning of the whole river ecosystem. It is now generally recognised that the whole flow regime is important for river maintenance (King *et al.* 2003) including:

- the dry-season low flows;
- the small floods that occur every year;

- the intermediate floods with occurrence intervals of two to five years;
- the rare, larger floods that can cause catastrophic disturbance to the system; and
- natural flow variability at daily, monthly, annual and inter-annual levels.

A direct link between flow and the distribution of riverine communities has been demonstrated on several rivers across southern Africa. For instance, the 1:2 year flood correlates with the lowest edge of the main band of woody vegetation and thus may be responsible for controlling encroachment of trees and shrubs into the active channel (Tharme and King 1998; King *et al.* 2000). The largest of the intra-annual floods similarly correlates with the dynamic band of pioneer species assemblages next lowest in elevation below the trees and shrubs.

Geomorphology

The physical structure of a river ecosystem is determined by the geomorphological processes that sculpt the channel. Channels can be either bedrock or alluvium. Channel geomorphology is dictated by the channel shape and the stability of the bed and banks (Rowntree 2001). The channel geomorphology in turn determines the substratum conditions and associated flow conditions (for any given discharge) that together create habitat for river fauna and flora. The river cannot be understood in isolation from the surfaces of the catchment. Rivers adjust their morphology (width, depth, slope and planform) to carry water and sediment supplied from the drainage basin. A geomorpholgically graded profile is one where the gradient of the river is gradually adjusted, and thus the ability to transport the available sediment load changes. As sediment is transported downstream, the size of the bed material declines as the channel gradient decreases, reflecting a balance between energy and material (Newson *et al.* 2002). Rowntree and Wadeson (1999) developed a hierarchical geomorphological classification system for South African rivers that encapsulated these downstream changes. This was subsequently updated (Rowntree 2001). In this classification system, river channel zonation is assessed in terms of gradient, valley form and characteristic channel features, such as substratum and reach types. The classification divides a river longitudinally within its catchment through a nested hierarchy. Each level of the hierarchy reflects the changing physical nature of both the river bed and the banks. Upper reaches usually consist of steep rivers in narrow valleys with poorly developed riparian zones. The width of the river and associated habitats increases with increasing distance downstream as valleys widen, resulting in less lateral constraint of river flow. Discharge increases and the system changes from one dominated by sediment supply in the headwaters to one of sediment transport in the lower, with equilibrium between these two processes occurring in the middle reaches. Flood terraces and floodplains become a more obvious feature of the riverine landscape as one moves downstream, with a consequent increase in the extent and complexity of the riparian vegetation.

Longitudinal changes to morphological units and hydraulic biotopes

In the geomorphological hierarchy of Rowntree (2001), the catchment is the coarsest spatial level that changes over the longest time span. The subsequent smaller levels, which change over shorter time spans, are segments, zones, reaches, morphological units and hydraulic biotopes. The latter two, are discussed further as they occur at the same spatial and temporal scales as the habitat investigations conducted for this thesis. The hierarchy of levels is nested and so each higher level imposes constraints on the features of lower levels. All levels, other than the lowest level of hydraulic biotopes, are defined in terms of their geomorphological characteristics and are relatively stable through ecological time. Hydraulic biotopes, on the other hand, are described with the addition of associated flow types and are thus more ephemeral than higher levels of the hierarchy (King and Schael 2001). This hierarchy is useful to understand the change in physical attributes of the river system down the longitudinal continuum from source to mouth. Sediment calibre changes down the river's length: large boulders and cobbles in the headwaters give way to gravel, sand, silt and clay in the lower reaches. These changes are reflected in the structure and presence of characteristic morphological units and associated hydraulic biotopes.

In this thesis, the description by Rowntree (2001) was used to determine patterns of longitudinal zonation. A short description of expected changes to morphological units and associated hydraulic biotopes down a longitudinal profile of a hypothetical river system follows. For a fuller description, particularly for the higher levels that are not considered here readers are referred to Rowntree and Wadeson (1999) and Rowntree (2001) who provide the basis for the short summary given next, unless otherwise indicated.

Morphological units

Morphological units are considered to be the building blocks of river channel habitats. There are three main groups of morphological units that may be either erosional or depositional features: pools, hydraulic controls and bars. Pools are erosional features with a relatively low width:depth ratio. Their macro-flow characteristics are controlled by a downstream morphological unit, the hydraulic control. This hydraulic control is situated where local steepening of the reach profile occurs. Thus, macro-scale flow of hydraulic controls is not determined by the position of downstream morphological units. Hydraulic controls have a relatively high width:depth ratio when compared to pools. They may form where large particles are deposited, such as in cobble riffles or boulder rapids, or may be erosionally resistant features such as bedrock rapids. Bars are the third group of morphological units and are depositional features that occur in a number of locations, such as on channel margins, within pools or even within hydraulic controls. They are often relatively mobile, consisting of small calibre substrata and therefore are short-term sediment stores.

Hydraulic biotopes

A morphological unit is composed of one or more hydraulic biotopes defined as '...spatially distinct instream flow environments...' and characterised by specific hydraulic and substratum attributes (Rowntree and Wadeson 1999). Hydraulic biotopes represent the lowest level of the geomorphological hierarchy and are defined in terms of visual characteristics of flow and the underlying substratum. Since they contain a description of flow that is used to describe differences between various hydraulic biotopes, the assemblage of biotopes present in a particular morphological unit is dependent upon discharge and thus temporarily unstable, changing with time in both type and proportion. Hydraulic controls are generally more diverse in the number and type of hydraulic biotopes present, when compared to pools that are more homogeneous.

The concept of hydraulic biotopes combines flow hydraulics and substratum conditions to describe the physical habitat available to, and experienced by, biota. The assumption is that flow and substratum characteristics interact to produce a physical environment (habitat) at the scale of approximately $1m^2$, which is relevant to small aquatic biota. Surface flow type, a substitute used to delineate the hydraulically complex conditions of hydraulic bipotopes, is used to identify them in the field (Newson and Newson 2000). Surface flow type is independent of scale and so may be applied equally to large or small rivers. Although bed conditions have a direct effect on surface flow, the description of surface flow types does not implicitly distinguish between different substrata. Substratum size class needs to be considered in its own right (Rowntree and Wadeson 1999).

Habitat mapping

A new field of science, variously called hydraulic stream ecology, ecohydraulics, habitat hydraulics, or similar, has emerged to study biotic/abiotic links and to develop a predictive capacity of how riverine biota respond to physical habitat changes (Rowntree 2001). Invertebrate physical habitat, in this thesis, is the biotic living space defined by the spatial and temporally dynamic interaction of channel features and the hydrological regime. Methods of assessing physical habitat are needed variously for fishery enhancement schemes, river restoration projects and environmental flow assessments. There are different methods used over a range of scales (Maddock 1999). For example habitat mapping is a coarse level but rapid assessment whereas PHABSIM (Physical Habitat Simulation System, Bovee *et al.* 1998) is a more detailed modelling approach requiring information on microhabitat variations with flow. Habitat mapping involves delineating the shape, physical characteristics and local hydraulics of a river site at a biologically relevant scale. This incorporates a mixture of qualitative assessment and physical measurement to record the form of a river based on field observation (Bisson *et al.* 1982, cited by Maddock 1999). Mapping surveys involve walking the study site along the river, identifying mesohabitats, drawing their location and extent, and measuring their physical attributes (e.g. water depth, width and velocity). A classification of mesohabitats should be chosen before field work commences (King and Schael 2001) as it provides a snap shot of the overall river conditions during data collection and can guide selection of sample points.

2.3 Riparian zones

In general terms, a riparian area is an area of land directly influenced by the presence of a freshwater body. As such, riparian systems are associated with nearly all continental waters – lakes, rivers, wetlands, springs and estuaries. Riparian zones occupy a three-dimensional transitional area (Wilson and Imhof 1998), or are ecotones (Swanson *et al.* 1992), between aquatic and terrestrial ecosystems and serve as conduits for the exchange of materials and energy from the one to the other (Richardson *et al.* 2007). In this thesis, the phrase riparian zone refers to areas directly adjacent to the active channel of the river that support vegetation communities. Riparian vegetation communities are distinctly different to neighbouring terrestrial communities, and reflect a distributional relationship to the flow regime of the river. As such, the vegetation of active floodplains and terraces immediately adjacent to the river channel is included whilst submerged aquatic communities within the active channel are not. Two main features separate riparian ecosystems, in this sense, from other wetland ecosystems (Rogers 1995):

1. a linear form due to their close proximity to rivers; and
2. a hydrological connection to upstream and downstream areas, at least intermittently.

Riparian zones in the Western Cape

Despite extensive research in the Fynbos biome, its riparian vegetation has not been intensively studied until recently. There is no formal classification of riparian vegetation communities for the Western Cape or the rest of Southern Africa. A thesis, two recent papers and two management reports addressed Western Cape riparian vegetation communities. Sieben (2002) completed a phyto-sociological study of riparian vegetation communities in the upper reaches of five rivers in the Hottentots Holland Mountains near Cape Town. Prins *et al.* (2004) defined reference conditions for south-western Cape rivers. They identified four prospective plant communities that formed a continuum, with the two outer-edge communities differing in soil pH, reflecting underlying geology. They concluded that riparian flora were generalists occurring widely across the Western Cape. Galatowitsch and Richardson (2005) assessed recovery of riparian vegetation communities in the south-western Cape following the clearing of woody alien species and synthesized various descriptions of plant communities into a database of 350 Western Cape riparian species. They concluded that indigenous tree regeneration was very low following clearing, thus slow recovery is likely to result without restoration intervention. Reinecke *et al.* (2007) investigated recovery of riparian communities following the clearing of woody exotic species in the south-western Cape. They developed a model that predicts where characteristic riparian flora may occur laterally on the river bank related to increased elevation above summer low flow, as recovery progresses. Holmes (2007) investigated targets for ecosystem repair in riparian systems nationally across, the Fynbos, Savanna and Grassland biomes. She found in all biomes that riparian systems generally had high resilience to invasion, following clearing of exotic species, as long as the cover of exotic species prior to clearing did not exceed 75%.

Impacts of alien invasion on riparian zones

Biological invasions are the second largest threat (after habitat destruction) to global biodiversity (Richardson and van Wilgen 2004). Semi-arid riparian zones are particularly prone to invasion by alien plant species (Dye *et al.* 2001) because of the availability of water in otherwise dry areas (Versveld *et al.* 1998). They are frequently disturbed by floods (Richardson *et al.* 2007), which uproot plants and create new habitat patches (Versveld *et al.* 1998) through fluvial deposition and erosion of sediments. Floods may provide these sites with propagules, such as water-borne rhizomes, seeds, and vegetative fragments, which can be distributed longitudinally along the system and up the banks (Tickner *et al.* 2001). Being frequently naturally disturbed, riparian zones require sustained management in areas where invasive species occur (Richardson *et al.* 1997). The growth of invasive species often outcompete local species during primary succession.

One of the most frequently cited impacts of woody alien invasion on riparian zones is the excessive use of surface and ground water by exotic species when compared to the volumes used by indigenous riparian vegetation (Dye and Poulter 1995; Versveld *et al.* 1998; Calder and Dye 2001). In fact, the recovery of streamflow following clearance of exotics from river catchments was one of the primary motivating factors for the Working for Water programme (van Wilgen *et al.* 1998, see below). This was based on the premise that indigenous woodland or scrub uses less water than catchments forested with exotic species (Scott and Lesch 1996; Dye *et al.* 2001). Woody exotic plant species have additional impacts, such as altering the riparian canopy structure and the abundance and variety of species and growth forms present.

As the invasion of the riparian community ensues, an indigenous mixed community may be replaced by an exotic monoculture. This alters the flow, availability and/or quality of nutrient resources within biogeochemical cycles, trophic resources within food webs, and physical resources such as light, sediment, space or water. This in turn has been shown to alter the delivery of allochthonous material during times of the year that do not correspond with aquatic macroinvertebrates life cycles, as is the case for indigenous woodland species (King 1981). Exotic plant species that change the physical structure of the ecosystem in this way are sometimes called 'transformers' (Richardson *et al.* 2000) or 'ecosystem engineers' (Crooks 2002).

The Working for Water programme

Locally, research into these wider issues is being conducted through the Working for Water (WfW) programme. The Working for Water campaign was launched as a multi-departmental public works programme to tackle two problems, that of alien vegetation and unemployment. The departments involved were the Department of Water Affairs and Forestry (DWAF), the Department of Environmental Affairs and Tourism (DEAT) and the Department of Agriculture (DoA). The aims of the programme are to: enhance water security; improve ecological integrity and biodiversity; restore the productive potential of land and promote sustainable use of natural resources; and invest in the most marginalised sectors of South African society (Uys 2003). The trade-off was the relatively low-cost clearing when compared to that of developing additional water-supply schemes. Of equal importance were the restoration and conservation of biodiversity, and promoting job equity to enhance the quality of life of previously-disadvantaged peoples (van Wilgen *et al.* 1998). Whether or not the predictions of significant benefits arising from alien control can be substantiated and whether or not the threat of invasive alien species can be curbed remains to be seen (van Wilgen 2004). Projects were selected on the basis of the need to re-clear areas previously cleared, the impact on woody alien invasive species on regional water resources, the extent and distribution of alien plants, the level of poverty and unemployment and the potential for institutional partnerships. The most immediate danger to the success of this programme was the will among government and other funding agencies and landowners to sustain follow ups. Clearing exotic species is expensive and needs to be maintained since the alien plant seed banks are long lived and large, and require annual follow ups to maintain control on regeneration of alien stock.

Sampling riparian vegetation communities

It is notoriously difficult to separate real from apparent patterns in ecological studies. It is difficult to keep a balanced view of all factors and contending hypotheses. No single method of community analysis, theoretical or experimental, can be guaranteed to give useful results about community patterns (May

1986). All methods are based on theoretical and statistical assumptions and should be criticised within these boundaries. The best methods after all are those that allow for useful interpretation of the data. Data on the distribution of vegetation communities can be collected in two conceptually different ways; the aim of both is to identify and describe groups of individuals or samples on the basis of their species composition. The method most commonly employed in the Western Cape to date has been based upon the Zurich-Montpellier school of phytosociology and is alternately called the Braun-Blanquet method (Werger 1974). Other numerical methods of community classification (see Section 2.5) have been applied in a whole range of sciences (Kent and Coker 1992) including botanical avenues and particularly restoration studies of post-project recovery (Harris 1999).

In the Braun-Blanquet approach, recognisable plant communities and their location are determined *a priori* by visual assessment, purposefully avoiding areas judged to be transitional between the selected community types. Different sized relevés are used to sample vegetation of differing size (e.g. larger relevés for trees than for ferns). A cover-abundance scale is used to record the composition of species within each relevé. The BB relevés are re-organised into a phytosociological table such that species common across the sampled relevés are excluded as much as is possible from the described communities. Community descriptions are based upon key species occurrences through the phytosociological table. Since this is based upon specialist knowledge and a certain intuition, different researchers are likely to recognise different arrangements of vegetation assemblages in the data table.

Other numerical methods do not recognise plant communities or their location *a priori.* In contrast to th BB method, data may be collected systematically over a wide area, and data-reduction techniques used to identify assemblages based on the grouping of similar samples. Samples falling outside these groups of similar samples may represent transitions between groups, or they may be samples where too few species were present for a proper identity of the sample to be made. The groups of similar samples define the plant assemblages for the area under study (Kent and Coker 1992).

It was decided not to delineate the riparian zone *a priori,* as per the Braun-Blanquet method. The chosen method was to sample across the transitions between the aquatic, riparian and terrestrial communities, and then to separate to the flora into respective groups based upon the species present, as per the numerical methods. Specifically, CLUSTER analysis and Multi-dimensional Scaling ordinations were chosen for this purpose (see Section 2.5). These methods, if repeated should produce the same result consistently using the same data set.

2.4 Aquatic macroinvertebrates

Invertebrates are distributed amongst riverine habitats in relation to many factors that act over a number of spatial and temporal scales. The main components of ecosystems that are considered to influence aquatic organisms are: the flow regime, availability of physical habitat (channel form and substratum composition), water quality (e.g. conductivity) and energy inputs from the catchment (e.g. allochthonous material, Thirion *pers. comm.*). In this way, community structure reflects a continuous sorting process through various environmental filters at regional or catchment scales, such as speciation, geological history and climate, to individual patches where local predation, substratum porosity and current velocity interact (Malmqvist 2002). Movement of invertebrates between lentic, lotic and terrestrial systems affects ecological processes across the boundaries between these systems. Invertebrates, for example, affect the cycling of carbon and other nutrients by virtue of their linking primary producers and consumers (detritus pools) to predators higher in the food chain.

At a local site scale, physical habitat heterogeneity influences the distribution of invertebrates. Some species prefer areas dominated by sediment deposition, such as pools, while others prefer erosion dominated areas, such as riffles. Generally, the coarser substratum of riffles provides more complex environments favouring the growth of moss and algae (Matthaei and Townsend 2000). Pools tend to have weaker currents, finer substratum and a gentler gradient (Townsend 1989). They are favoured by sediment burrowing organisms. Riffle invertebrates tend to be scrapers and suspension feeders that obtain food on the exposed surfaces of substrata. Life within the finer sediments, on the other hand, offers biofilm and detritus as food as well as protection from fish predators (Soluk and Collins 1988). Many invertebrates take advantage of other riverine habitats, such as alongside the channel in side-arms, backwaters and floodplains (Stanford and Ward 1993; Ward and Tockner 2001). Many aquatic invertebrates leave the river for mating and dispersal and thus serve as food for birds, mammals and other invertebrate predators (Jackson and Resh 1989).

Adult (aerial) and larval (aquatic) stages of riverine macroinvertebrates disperse over small or large distances. Dispersal may take place downstream via drift, or upstream and laterally through crawling, or flying, in order to find mates, food or to colonise other areas. For example, blackfly eggs are deposited in large aggregations with the newly hatched larvae dispersing downstream (drift) in a density-dependent fashion (Fonesca and Hart 1996). Limnophilid caddis fly larvae and large stoneflies tend to disperse via crawling rather than by drift. Freilich (1991) found that stoneflies aggregate in riffles and avoid silty pools. Bunn and Hughes (1997) found that the genetic variation between populations of gerrid bugs, caddis flies and mayflies in a hierarchy of rivers was higher in different reaches of the same river than between rivers from different neighbouring catchments. This highlighted the possibility of pronounced site-specific community dynamics due to local biotic community processes, such as predation, competition and disease (Malmqvist 2002).

Macroinvertebrates and disturbed river systems

The South African River Health Programme (RHP, www.csir.co.za/rhp/index.html), implemented by the Department of Water Affairs and Forestry's (DWAF) Institute for Water Quality Studies (IWQS) in January 1995 (Murray 1999), makes use of biological indicators (e.g. fish communities, riparian vegetation, aquatic macroinvertebrate fauna) to assess the condition or health of river systems. The rationale is to provide an integrated and holistic measure of river health. The goal of the RHP is to serve as a source of information regarding the ecological state of river ecosystems in SA, in order to support rational management of freshwater as a natural resource. The use of abiotic and biotic attributes of rivers to assess river health has lead to the belief that the fauna and flora of one river reach, as defined by a number of measurable physico-chemical characteristics will be present in another reach with comparably similar characteristics. This generally held view that rivers with the same features will have the same biota has been shown to be simplistic by King and Schael (2001) who tested it at a variety of spatial scales in Western Cape rivers. They predicted that mountain stream communities of different rivers would have similar species of invertebrates due to their physical similarity. In contrast they found that mountain streams and foothill zones from the same river had more similar invertebrate communities, than with corresponding zones on other rivers. They concluded that invertebrate communities in the same zones from different catchments were not necessarily similar and coined the term 'catchment signatures' for this phenomenon.

A study of invertebrate species traits (Townsend *et al.* 1997) in relation to temporal and spatial heterogeneity showed that communities suffering from anthropogenic disturbance had higher percentages of individuals with the following certain traits. These were small size, high adult mobility, habitat generalist (postulated to confer resilience in response to disturbance events), clinger, streamlined/flattened and multivoltine (postulated in resistance to disturbance events). Other resilience traits were early age of reproduction, short reproductive cycles, potential for regeneration and the ability to colonise from refugia, while some other resistance traits were firm attachment, streamlined body form and invulnerable life stages.

Stone and Wallace (1998) recommended using a wide range of measures to assess long-term aquatic invertebrate recovery, such as macroinvertebrate abundance and biomass, secondary productivity, taxon richness and functional diversity. Their method compared differences in invertebrate taxonomic groups based upon percentage similarity matrices constructed from habitat-weighted abundances in both reference and disturbed rivers and made use of cluster dendrograms (see Section 2.5) to portray the relationship between reference and disturbed groups (Brower and Zar 1984).

Methods of bioassessment and the South African Scoring System (SASS)

There is growing interest in rapid bio-assessment techniques (van Eeden 2003) since the tentative starts in the early 1970s when Chutter (1972) first developed a biotic index based on species diversity and abundance of aquatic macroinvertebrates. This was improved upon and resulted in the South African Scoring System (SASS, Chutter 1998). The latest version, SASS5 (Dickens and Graham 2002), now forms the backbone of the River Health Programme in South Africa.

The success of bio-assessment techniques to assess river health is widely recognised. This is, in part, due to the presence of a wide range of aquatic species, each reflecting different environmental conditions. Benthic macroinvertebrates are valuable in this regard, due to their ease of identification to family level with the naked eye, rapid seasonal life cycles and their largely sedentary habits. As such, aquatic biota integrate and reflect on cumulative effects that impact on an ecosystem over time (Ollis *et al.* 2006). According to Dickens and Graham (2002), the SASS5 method was designed for low to moderate flow conditions in lotic systems only and works best when the diversity of biotopes is wide and includes riffles and pools. They recommend interpreting SASS5 data in relation to the availability, diversity and quality of habitat, and more broadly in relation to season and catchment specific factors. There is likely to be natural variation during the year, between years and across eco-regions. Steps in the SASS protocol, as described by Dickens and Graham (2002) are summarised below.

SASS5 method description

Invertebrates are collected in defined biotopes *viz.* stones in current (SIC); stones out of current (SOC); marginal vegetation in and out of current (MVegIC and MVegOC); aquatic vegetation (Veg) and gravel, sand and mud biotopes (GSM). During sampling, approximately one minute of 'hand picking' for specimens that might be easily missed is carried out, such as snails and fast moving pond skaters, or burrowing organisms in very thick mud. The sample is washed out of the net and into a sample tray that is topped up with clean water. Separate trays are used for the six categories of biotope. Debris and stones are removed before identification and are shaken to dislodge attached organisms. Samples may be preserved for laboratory identification at a later stage if necessary. During identification, organisms are listed in families on the SASS scoring sheet, where a sensitivity score for each taxon reflects the susceptibility or resistance to pollution. Lower scores indicate greater resistance while higher scores represent greater susceptibility. Viewing is done for a maximum of 15 minutes per biotope but may stop earlier if no new taxon is seen after five minutes. The abundance of organisms is indicated from 1, 2 to 10 (A), 10 to 100 (B), 100 to 1000 (C) and >1000 (D) on the data sheet. Three principle results are calculated; SASS score, Number of Taxa and average score per taxon (ASPT). SASS score is calculated by summing the quality score assigned for each taxon on the score sheet. The number of taxa found is divided by the total SASS score to provide the ASPT. While separate results may be calculated for each biotope, only the result calculated from the Total column represents the SASS5 score for the site. In principal, there will be less biotic diversity where habitat diversity is low, such sites will have a lower SASS score. The ASPT is less affected by habitat diversity and is thus a more reliable measure of river health (Dickens and Graham 2002).

2.5 Numerical classification

The goals of numerical classification (Clarke and Warwick 2001) are to group a set of individuals, quadrats or vegetation samples into classes on the basis of their floristic attributes. The numerical methods rely on a set of rules dictating how the groups of individuals or samples are arranged and so, for any one set of data, any researcher using a particular method should produce the same result.

Two grouping methods have been used for the analysis of vegetation and macroinvertebrate data in this thesis. These are CLUSTER analysis and non-metric multi-dimensional Scaling (MDS) ordination. Both these two analyses aim to group samples that are similar together such that samples within a group are more similar to one another than to samples in different groups (Clarke and Warwick 2001). Ideally each group, whether revealed from CLUSTER or MDS ordination analyses, should contain samples with a similar species composition.

Cluster analyses

Clustering (Clarke and Warwick 2001) is a technique that has widespread utility in ecological studies. All classification techniques are arbitrary and thus should be performed in conjunction with a range of other methods such as ordination or statistical testing, to obtain balanced and reliable conclusions. Methods of hierarchical agglomerative clustering, chosen for use in this project, are the most commonly used clustering techniques. Data are reduced to a similarity matrix from which samples are fused into groups, starting with the highest mutual similarities and ending with a sample matrix that contains all the samples. The results are portrayed on a dendrogram, or tree diagram, with the x axis representing the full set of samples and the y axis defining the level of similarity. The analysis attempts to group samples into discrete clusters and not display their inter-relationship on a continuous scale (Clarke and Warwick 2001). Cluster analyses are seen as being useful to determine 'true groups' in classifications (Manly 1992).

Ordination of samples

In contrast to cluster analysis, ordination shows the inter-relationships between samples on a continual scale. It is widely recommended that it is performed in conjunction with cluster analysis since agreement between the two methods strengthens the belief and adequacy of each. There are a number of ordination methods. The earliest was the Bray & Curtis polar ordination (PO) followed by Principal Components Analysis (PCA), reciprocal averaging (RA), correspondence analysis (CA) and then detrended correspondence analysis (DCA). Canonical correspondence analysis (CCA) is another method, differing from those already mentioned in that it incorporates the correlation and regression between floristic data and the environmental variables within the ordination, whereas the other methods (PO, PCA, RA, CA and DCA) are all indirect methods that produce the ordination independently of the environmental data. In these indirect methods, the explanatory environmental variables are than superimposed onto the ordination afterwards. The mathematical algorithms for all these methods vary in their complexity and numbers of assumptions made regarding the relationship between samples and the nature of the data. Kent and Coker (1992), Clarke and Warwick (2001) and Manly (1992) provided detailed descriptions and comparisons between the different methods.

2.6 River restoration

Restoration is the most common term used to describe rehabilitative management of freshwater ecosystems. Restoration practitioners have either described their activities following a Balance or Flux philosophy (see section 1.1). Flux proponents used the term 'restoration' to describe any sort of intervention that aims to improve the current state of a degraded ecosystem. Balance proponents were strict in determining whether an intervention was a restoration, rehabilitation or remediation/enhancement, based upon the degree to which a particular ecosystems state was restored back towards a historically natural condition.

Following a **Balance of Nature (BON)** view (Rutherfurd *et al.* 2000), human impacts on ecosystems are seen as a non-natural disturbance. Systems used on a sustainable basis by humans thus cannot be restored, because the impacts associated with that use change the ecosystem from its pre-disturbance state. Restoration under the BON paradigm aims to return the degraded or impacted ecosystem to a historically natural state. This is taken to be the state prior to intensive human settlement. River restoration may only be achieved if, "...the entire stream network and most of the catchment surface are also restored".

BON proponents define other terms that differ in the degree to which a restored state is achieved. These are compared by King *et al.* (2003) and include the terms rehabilitation, remediation and enhancement. Rehabilitation is described as '...the returning of an impacted ecosystem toward its pre-disturbance state...' acknowledging that neither the entire river network nor the greater part of the catchment can be rehabilitated. Where rehabilitation is not attainable Rutherfurd *et al.* (2000) offer remediation as a suitable alternative. Remediation, after Calow and Petts (1994) and Petts *et al.* (2000), is defined as "...improving the current state of an ecosystem without reference to its initial state." Remediation or enhancement may be attempted in order to mitigate the effect of disturbance and to provide optimal conditions for a highly valued species such as game fish (Petts *et al.* 2000).

Flux of Nature (FON) proponents (Calow and Petts 1994; Jackson *et al.* 1995; Meier 1998; Fogg and Wells 1998; Petts *et al.* 2000) define restoration to be, "...an attempt to bring the river back to as high a level of ecological integrity as possible, taking into account the prevailing socio-economic, political, and technological constraints." They do not believe ecosystems can be recreated to resemble their pre-disturbance conditions since these are continually changing anyway. The goal is to create a self-sustaining ecosystem, which does not degrade further and therefore requires no intervention to maintain the restored state. Restoration is used as an umbrella term encompassing all the different levels of intervention, such as rehabilitation, remediation and enhancement. Using the term restoration loosely in this way means the objective of each restoration activity may differ depending on the degree to which a prior state is achievable.

Principles of restoration

Generally, restoration principles focus on management principles, such as ensuring objectives are realistic in terms of budget, or that stakeholders are involved. Such principles are vital for restoration success, but nested within them should be a sub-set of ecological and geomorphological principles that will guide scientific aspects of the work. Without these, restorations stand the risk of being random experiments that will make little or no contribution towards scientific advancement or technical expertise (King *et al.* 2003). The emerging global weakness of river restoration has been caused by an over-emphasis on partial or physical aspects, a lack of planning and structured processes, a paucity of available ecological, historical and reference data on which to base restoration planning, the meagre attention to scientific process and principles, and a lack of evaluation of outcomes (Uys 2003). In South Africa, concerns about river restoration (Uys 2003) are:

- a lack of understanding of what restoration ecology involves;
- the paucity of research attention and funding directed towards it;
- the inadequate attention of individual projects to planning and the environmental outcomes of the work;
- the lack of a scientific basis for restoration; and
- the dominance of traditional engineering approaches rather than bio-engineering techniques.

River restoration in South Africa

The majority of South African restorations to date have been typically *ad hoc,* and thus have not contributed conceptually towards the slowly growing field of expertise. Many restorations are undertaken by private companies, consultants or community groups and are not reported in accessible media: the learning is lost. The term *rehabilitation* is contemporary and fashionable and thus is attached to many projects which in fact have very little to do with reinstating naturally functioning ecosystem processes. These projects are more accurately described as 'river improvement', 'stormwater control', or 'greening of urban areas'- all of which have value in themselves but are not restoration (Uys 2003). A centralised agency that will coordinate local restoration projects is needed to collate and disseminate data and knowledge.

In this thesis the term **restoration** has been used, as per the Flux paradigm, to describe all activities associated with improving the state of degraded river ecosystems.

Presently there is no such programme coordinating river restoration in South Africa. The Working for Water campaign (see Section 2.3) and the River Health Programme (see Section 2.4) will serve as excellent partners to a national river restoration programme (Uys 2003). Further developments towards co-ordinating river restoration research locally are needed before such a national centre of excellence on river restoration could be developed. River restoration in South African cannot move forward until local authorities start communicating successes and failures with one another and with restoration scientists in an open forum. Despite a lack of funding, many projects are being conducted and lessons therein can benefit future projects.

2.7 Ecosystem recovery

The functioning of river ecosystems and the multiple and continuing effects of disturbance events are well described in limnological literature (White and Pickett 1985). Less so is the process of recovery following a disturbance, be that by anthropogenic or natural means. The process of recovery is fundamental to restoration ecology and is dictated by all aspects of an ecosystem's response to disturbance. This may be monitored through ecosystem indicators, such as biological, geomorphological and hydrological components of river ecosystems. Following disturbance, different parts of the river ecosystem mosaic recover their structural and biologically complexity in different ways and over different spatial and temporal scales. During this process of recovery, the state of the river and all its associated ecosystem components may not necessarily resemble the original river's state. One of the difficulties confronting restoration ecologists is assessing whether or not a particular ecosystem (or site) is recovering, and if so, where along the trajectory of possible recovery states it lies. The first suggested step in a restoration plan is to assess this by asking (after Uys 2003):

- whether the system is in a state of recovery following the disturbance;
- how recovery will proceed and over what time period the extent of these effects will be felt; and
- how the system will respond to interventions planned to assist the recovery process.

These assessments require at a minimum the expertise of experienced fluvial geomorphologists, hydrologists and freshwater ecologists. The process normally includes acquiring information on the pre-disturbance state and flow records, the site's history and past disturbance events, catchment landuse and causes of degradation (Uys 2003). Decisions are made on whether to assist by accelerating the process of recovery, or to allow natural recovery to proceed un-assisted (see below). Implementing and co-ordinating a restoration project is a complicated and lengthy process that is well described by Rutherfurd *et al.* (2000), has been summarised by King *et al.* (2003) and was tested locally by Uys (2003). Readers interested in managing restoration projects and the multi-party stakeholder natures of these projects are referred to these texts for further information.

Spontaneous succession

Spontaneous succession (or unassisted natural recovery) as a restoration technique is usually applied when resources are limited. It relies upon the local supply, and natural dispersal mechanisms, of biota. The *treatment* as such is to allow natural processes of recovery inherent in the system under investigation, to operate without anthropogenic interference. Advantages and disadvantages of relying on spontaneous succession (Aksey-Doran 1999) include:

Advantages

- savings in labour and cost;
- regenerating and dispersing colonists establishing more easily than transplanted/transported individuals; and
- stable communities that are diverse in composition and structure and within current ecosystem thresholds and processes.

Disadvantages

- reliance upon nearby sources of colonists, either from undisturbed areas upstream or survivors at the restored site (for example soil/canopy stored seeds);
- regeneration may be patchily confined to one river bank or occur as patches on alternate river banks, exacerbating erosion; and
- continued maintenance of follow-up treatments, such as clearing of exotic vegetation.

Restoration success ultimately depends on whether populations, communities and ecological functions attain limits typical of reference systems. There is increasing evidence that the removal of stressors is not always sufficient to guarantee restoration success. The removal of anthropogenic stress coupled with solving physico-chemical problems does not guarantee the recovery of impacted organisms, communities or ecosystem processes (Ormerod 2003) since other factors may prevent recovery from taking place.

All ecosystems will eventually recover following a disturbance provided that (1) the factors causing damage or constituting a disturbance are eliminated, (2) there exists a suitable supply of colonising species, and (3) the physical environment matches the needs of the species (Frid and Clark 1999). Natural recovery processes, however, may take longer than current value judgments allow hence earlier recovery through active restoration may be deemed appropriate. Because of the open nature and lentic connectivity of rivers (section 2.1), recreating physical habitat structure may in some cases be sufficient for natural re-colonisation to proceed. Therefore all schemes should make an assessment of the available nodes of propagule supply, which harbour adults of the target species that may supply propagules, and of the existence of transport corridors through which the propagules may be transported. Restoration is usually more complex than simply re-creating physical habitat. It is impossible to transplant a complete functioning ecosystem (Frid and Clark 1999). Focussing only on biologically important species, such as vegetation, or other ecologically important key species, may result in transient success if the transplanted individuals die and are not replaced by internally operating mechanisms. Incomplete knowledge of river community structure results in important biological or physical components often being overlooked or not being known. Thus restoration requires stimulating naturally operating ecosystems by considering spatial relationships, recruitment rates, trophic interactions and population and community dynamics. This is difficult in practice due to a lack of data and paucity in current ecological science where the principles of restoration are concerned. As a result it is difficult to accurately predict trajectories or paths of ecosystem recovery and then develop assessment criteria for these. As restoration projects are assessed globally the phenomenon of alternative stable states continues to recur. The study of alternative states involves investigating drivers, both physical and biological, which control the trajectory of recovery of the ecosystem under investigation (Suding *et al.* 2004). Restoration often fails when attempts are made to restore historical abiotic conditions, without considering current biotic factors and the possibility of positive feedbacks between abiotic and biotic factors. As ecosystems evolve, changes occur continually as connectivity and organisation changes, native species are lost, shifts in species dominance occur due to trophic interactions and/or alien invasions, and biogeochemical cycles are disrupted. Models of alternative stable states incorporating ecosystem thresholds and feedbacks help address these constraints.

The 'state and transition' model of ecosystem recovery following degradation proposes that linear trajectories between ecosystem states do not exist, rather, disjuncts or transitions exist between different ecosystem states (Perrow and Davy 2002). These alternative states are controlled by abiotic or biotic factors that create thresholds, which need to be overcome in order to move from one state to the next. Should an undesirable state persist, a recovery trigger may 'kick-start' a trajectory towards a more favourable state (Askey-Doran 1999). Recovery triggers may be controlled by colonisation bottlenecks, where small interventions may remove the obstacle. The concept of minimal intervention recognises that interventions are acceptable but minimised, and are only implemented to the extent necessary in order to set the path towards a new trajectory. Thus restoration practitioners must acknowledge the existence of alternative stable states and be prepared to gear management to disrupt feedbacks that constrain the recovery trajectories (Suding *et al.* 2004).

In this thesis, the role of spontaneous succession as a method by which recovery may take place is investigated. Specifically, it is questioned whether without direct human intervention, ecological change following exotic vegetation clearance in the Silvermine River catchment will result in riparian vegetation and river macroinvertebrate assemblages becoming more similar to those in neighbouring, undisturbed rivers.

3. STUDY AREA

3.1 The Silvermine River catchment

The source of the Silvermine River is at an altitude of 640 m at 34°07'S and 18°27'E, 10 km south of Cape Town (Figure 3.1). The river derives its name from attempts, in the late 17th century, to extract silver from the catchment (Cobern 1984). It is a short, naturally perennial river approximately 12 km long, with a catchment area of *ca.* 21 km^2. The winter rains deliver an annual runoff of 4.5x 10^6 m^3 a^{-1} at a reservoir (Figure 3.2) in the upper reaches (Heinecken 1982). Highest rainfalls typically occur from June to August, with February being the driest month (Appendix 1). Rainfall in the upper catchment is in the order of 1200 mm a^{-1}, whilst the lower slopes are much drier with a mean of 600 mm per annum at the river mouth (Heinecken 1982). The upper reaches consist mainly of quartzitic sandstones of the Table Mountain Group, which overlay shales of the Malmesbury Series that are highly leached in the rocky outcrops (Heinecken 1982). A high-altitude wetland close to the upper end of the catchment feeds into a reservoir. In the middle valley, the lower mountain slopes consist of wind-blown Aeolian sands and shelly material, which overlay weathered Cape Granite (Heinecken 1982). These were blown up the valley from the lower-lying sand dunes, forming a deeper mixed soil. The lower catchment has marshes bordering the river, with organically rich, sandy materials (Harding 2000).

River flow

The reservoir in the upper reaches was constructed in the late 19th century to provide water for Kalk Bay and Muizenberg and has a capacity of 83,000 m^3 (Heinecken 1982). An access track used to surround the high-altitude wetland and numerous seeps draining toward the track were diverted into storm-water drains. These changed the flow pattern into the wetland from general seepage to small channels that have eroded the peaty soils. Use of the track was discontinued toward the end of 2001 to allow the wetland vegetation to re-grow. Water flows over the reservoir spillway after the winter rains start, but little moves past the dam and down the river in summer. This is because river flow is abstracted from the upper reaches out of the Silvermine catchment. Most flow is stopped by the dam. There is some seepage from the reservoir that is diverted into the old Kalk Bay/Muizenberg pipeline, along with water that has been piped underground from the reservoir to the diversion weir, 1 km downstream. Varying amounts of this water are used by Westlake Golf Course (WLGC) and the remainder empties into Sandvlei coastal lake. A detailed description of this abstraction, the resulting pattern of surface flow in the catchment and the associated impacts are summarised in Appendix 1 (Reinecke *et al.* 2003).

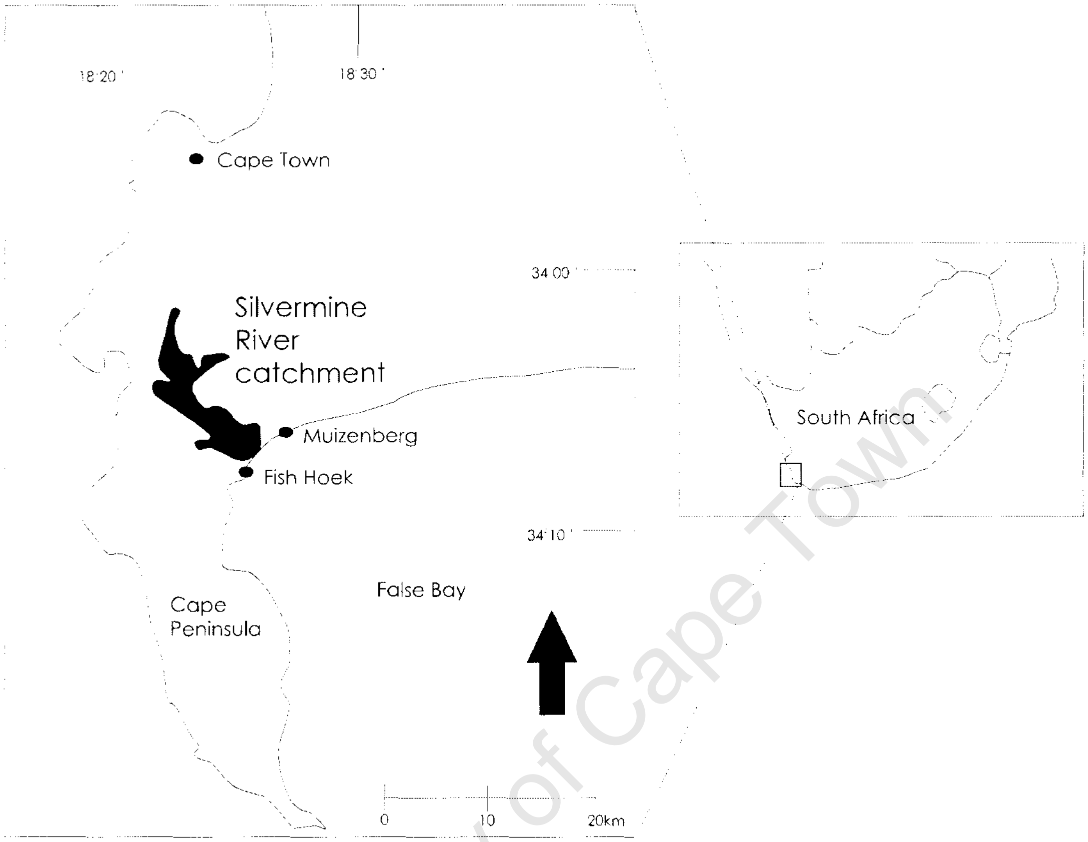


Figure 3.1 **Location of the Silvermine River catchment on the Cape Peninsula.**

The study showed that surface flow is reduced to almost nothing during the dry summer months from the abstraction weir down to the junction with the first tributary downstream, where surface flow comes into the main channel from another section of the catchment. This 2 km section of the main river channel in the upper catchment remains void of flow during summer low flow until the winter rains begin in late autumn and flows are able to overtop the abstraction weir (Figure 1, Appendix 1). Five tributaries enter the main channel at various points, three joining the left bank and two the right bank (Figure 3.2). Tributary 5, situated lowest in the catchment, is only known from maps; the channel, if it still exists, was not found. In the lower reaches, several houses have been built in the coastal floodplain and a few actually incorporate the river channel into their gardens. Some residents tap the aquifer of the floodplain via boreholes. The Clovelly Country Club (CCC), a golf course, is situated adjacent to the same floodplain. CCC taps river water during winter into storage ponds for irrigation water during the drier summer months.

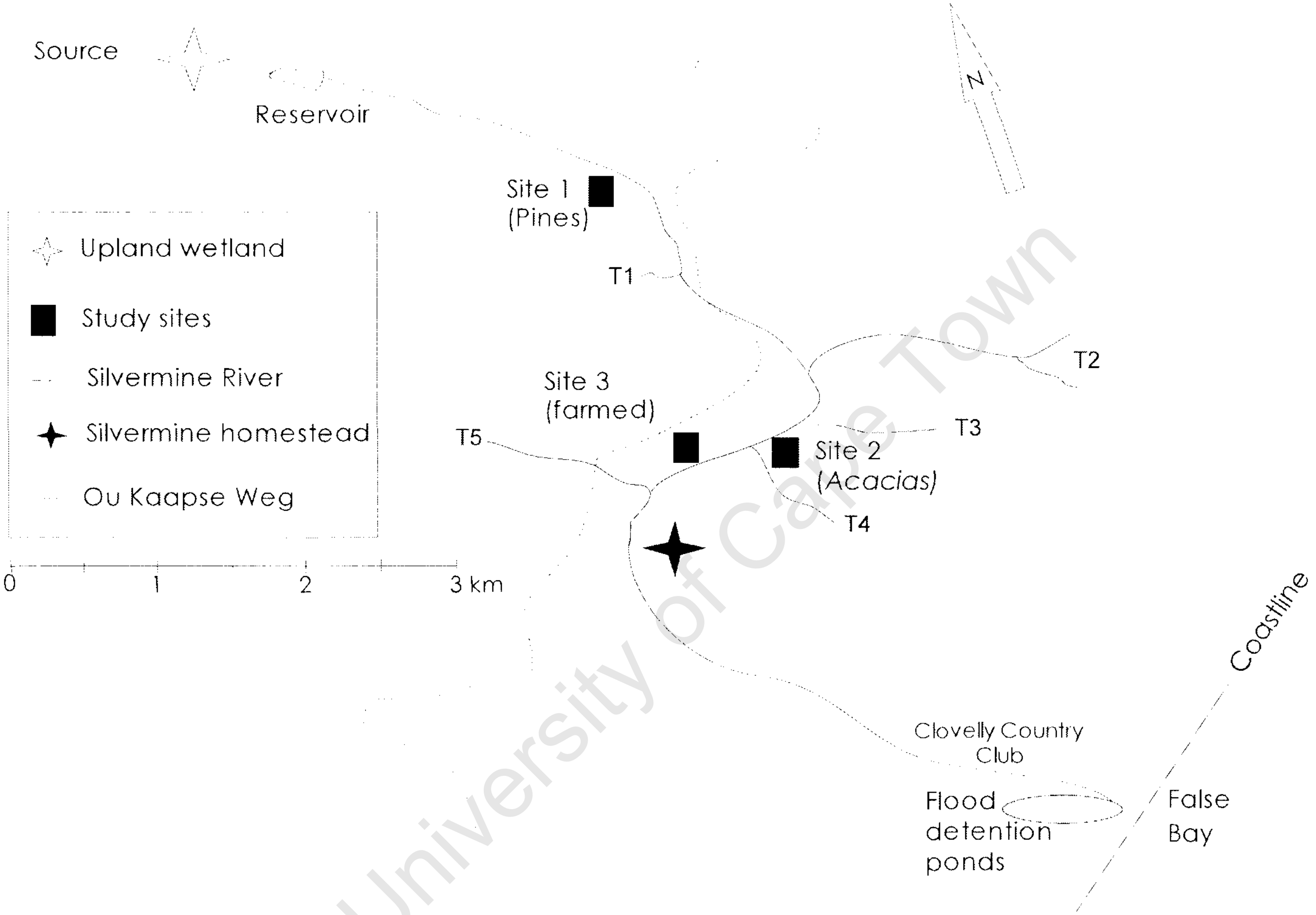


Figure 3.2 **The Silvermine River catchment.** The catchment is bisected by the road Ou KaapseWeg into the upper reaches and the lower valley sections. The location of the three study sites and their main relevant characteristics are shown along with other features of interest. The entire catchment upstream of the Silvermine homestead was burnt in the fires of January 2000. T1-5 = tributaries.

There is some indication that the river course may have been changed and restricted in order to increase the extent of the golf course fairways and greens. Downstream, urban areas encroaching on the coastal floodplain are prone to flooding during peak rains. In order to combat this, three flood detention ponds were built increasing the width to depth ratio of the lower Silvermine River considerably for a distance of approximately 1 km upstream of the river mouth. There are no serious problems of water pollution in this catchment, due to the lack of urban, agricultural and industrial areas. The semi-urban areas in the lower reaches must have some impact on the river, but the extent of this is not known (Harding 2000).

Recent landuse and catchment disturbances

Three study sites were chosen to represent a diversity of alien vegetation and to span the upper, middle and lower catchment (Figure 3.2). The sites differed in the severity of invasion and species of alien vegetation present. Extensive mountain fires burnt much of the catchment's vegetation in January 2000 incinerating most of the standing vegetative biomass: all areas upstream of the Silvermine Homestead were burnt (Figure 3.2). Areas previously covered by thick *Acacia* spp. and pine forest were laid bare. After the fires, new fire policies were implemented by the Table Mountain National Park (TMNP), which controls activities in the catchment. These included the implementation of an alien-eradication programme for the remaining and re-generating alien species. Clearing of alien vegetation from all surfaces of the catchment took place in small blocks through a tendering process, managed by the TMNP staff. As tenders were met and approved, catchment blocks were cleared. As a result, through the course of this project, the whole catchment developed a mosaic vegetation pattern as different blocks were cleared and subjected to follow up clearings. Post-fire recruitment of alien seedlings differed between the three study sites, and thus each was subject to a different clearing regime.

Site 1, upstream of Ou Kaapse Weg in the upper reaches, was afforested with mature *Pinus pinaster* (Cluster Pine) some of which were not burnt during the fires. These remaining adults were subsequently cleared towards the end of 2000 along with recruiting pine seedlings. The adult trees were cut off near the base and the root stock was left in place to limit unnecessary soil erosion, while the few germinating seedlings were hand pulled. Since pines do not coppice it was not necessary to treat the cut stumps with herbicide. All cleared woody biomass was removed. Roots and short stumps of the old trees remain visible, but few, if any, pine saplings remained after field work was completed in April 2002.

Sites 2 and 3 were located in the middle valley section of the catchment and presented a more difficult restoration problem. At Site 2, the main species present were *Acacia longifolia* (long-leafed wattle) and *Acacia saligna* (Port Jackson willow), the latter being far more difficult to clear as, prior to the fire, it was more widespread and had formed dense stands both on the catchment slopes and in the river channel. There was vigorous recruitment of *A. saligna* seedlings after the fire and Working for Water clearing teams initiated clearing activities in early 2000. Seedlings were left to grow for approximately six months before being removed. The larger trees that could not be removed were felled at the base and then poisoned with the herbicide Garlon 4© and a mixture of blue or red dye in order to distinguish treated stumps. The hacked or pulled plants were stacked near the clearing sites and burnt.

At Site 3, much of the valley floor was covered by *Pennisetum clandestinum* (kikuyu grass). This area had been used for the cultivation of vegetables and also contained terraced landscaped gardens. At the time of data collection, there were no plans to remove the alien grass, and at the last site visit (January 2005) the previously cultivated lands remained covered with kikuyu.

3.2 Longitudinal profile, zonation and location of study sites

The Silvermine flows from its mountainous source and thus has a steep gradient into the ocean over a short distance. Reinecke and King (2003) described four zones along the river's length using the hierarchical geomorphological classification method developed for South African rivers by Rowntree and Wadeson (1999) and adapted by Rowntree (2001). The three study sites, their co-ordinates and other key features used in assigning longitudinal zonation are given in Table 3.1. A short *mountain headwater stream* is followed by a long *mountain stream*, a lengthy *foothill*, a short *lowland* zone and the mouth (Figure 3.3). The change from mountain stream to foothill occurs at the junction of tributary three (Figure 3.2), just upstream of Site 2. Site 1 was located in the mountain stream zone while sites 2 and 3 were located in the foothill zone. Site 3 could not be located further downstream as no clearing was planned for the lower sections of the catchment over the data-collection period.

Table 3.1 **Location and zonal features of the three study sites.** Co-ordinates given are those on the left bank of the first cross-section at each site at the water's edge. Zonation based upon Rowntree (2001), adapted from Reinecke and King (2003).

Site #	Co-ordinates		Zone	Alien species present	Slope	Altitude (m asl)	Distance from source (m)
	Latitude	Longitude					
1	34°05′18″	18°25′8″	Mountain stream	*P. radiata*	0.07	320	3750
2	34°06′15″	18°25′16″	Foothill	*Acacia saligna*	0.04	135	5975
3	34°06′17″	18°24′47″	Foothill	*Acacia* spp., *P. clandestinum*	0.04	115	6525

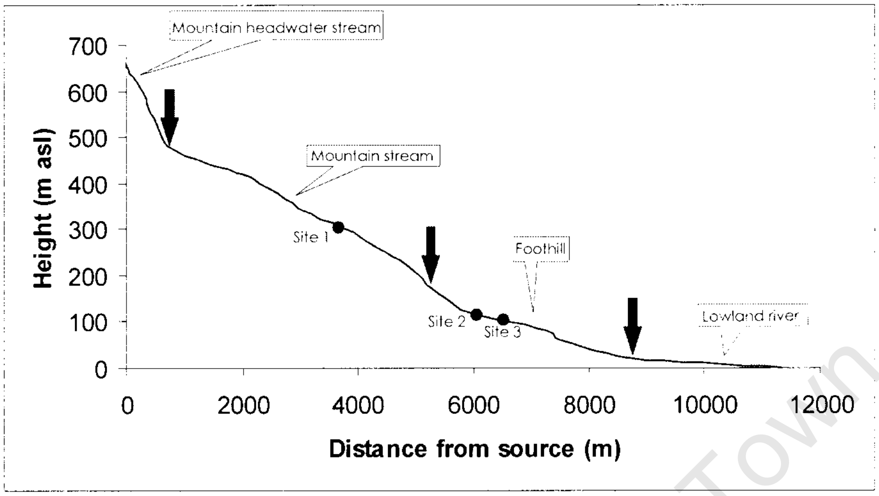


Figure 3.3 **Longitudinal profile of the Silvermine River** (Reinecke and King 2003). The locations of the three study sites are shown in their respective longitudinal zones. Black arrows indicate the transition from one zone to the next.

4. METHODS

Two main data sets were collected to meet the three study aims (Section 1.3): physical site characteristics (Aim 1) and biological indicators of recovery (Aims 2 and 3). The dates of the field trips and the data collected during each phase of this project are summarised in Table 4.1. The same set of data was collected at each site over two successive summers, 2001 and 2002, in order to make comparisons between years.

Table 4.1 Dates and purpose of field work conducted.

Data collected	Summer 1	Summer 2
PHYSICAL DATA		
Cross sections, channel outline surveyed	30 Nov to 15 Dec 2000	24, 25 Jan 2002
Aquatic habitats mapped	27 Feb to 13 Mar 2001	28 Mar to 12 Apr 2002
High water flood level marked	25 Jul, 1 & 8 Aug, 21 Sep 2001	
BIOLOGICAL DATA		
Aquatic macro-invertebrates collected	27 Feb to 13 Mar 2001	03 Apr to 12 Apr 2002
Vegetation surveys completed	29 Jan to 12 Feb 2002	18 Dec 2001 to 04 Mar 2002

4.1 Physical characteristics

Two different approaches were used to monitor and record changes to the physical structure of the aquatic habitats within the river channel. The cross-sectional channel shape and the dimensions of the channel outline were surveyed. In addition, two layers of an aquatic habitat map, representing the abundances of different flow and substratum types, were drawn onto the surveyed channel outline maps.

Surveying cross-sections and channel plans

Channel outlines were surveyed using an electronic theodolite (Leica TC307 model) and a standard Leica prism on a staff. These spot height data were used to produce plan maps of the channel at each of the three sites. The locations of the surveyed cross-sections were entered on each of the channel outline maps (Chapter 5). Three cross-sections were surveyed in at each of Sites 1 and 3 while five where surveyed in at Site 2, since it was longer. Cross-sections depict channel shape and may be used to calculate a channel's width:depth ratio. In this study the cross-section data was particularly important for recording the location of the vegetation plots (see Section 4.2) on the channel bank with reference to the wet edge of the river channel and to each other. The cross-sections, located at points deemed representative of different conditions at the sites, were numbered by site, and in sequence within a site from upstream to downstream (e.g. CS 1-1 is the most upstream cross-section at Site 1). Each cross-section extended through the active channel and beyond the macro-channel from the left to the right bank (when looking downstream) in a straight line perpendicular to the river flow. The cross-sections were re-surveyed twice, once during each year of data collection, in order to record the change in channel shape between sampling periods. The cross-sections were defined by three control points, usually two on the left bank and one on the right bank that were permanently cemented into the bank as control beacons. All the beacons were surveyed to a single master beacon at each site and each of these to the nearest trigonometric beacon located on the Steenberg plateau near the ticket office controlling access to the upper reaches and the reservoir. Surveying was done by graduate students at the Geomatics Department at the University of Cape Town: Mr MC Briers and Mr P Bornman. The surveyors used the Computer Aided Drafting (CAD) programmes Autocad and Allycad to generate the channel plan maps and cross-sectional profiles.

Aquatic habitat mapping

Maps were drawn of the aquatic habitat at all three sites in order to assess any changes in aquatic habitat over one year between the two sampling periods. The two components of hydraulic biotopes (flow and substratum types) were used to locate replicate 'habitat types', represented by different combinations of flow and substratum types at each of the study sites and thus to guide where invertebrate samples may be taken. The assumption behind this methodology was that different invertebrate assemblages would be located in different flow/substratum combinations. Further, the maps were to serve as a record of the flow and substratum types at the time of sampling. The methods used by King and Schael (2001) were followed to record different flow and substratum types. In their definition of hydraulic biotopes, eight categories of substratum and 14 categories of flow types were used. These were reduced (in this thesis) to six categories each of flow and substratum types, and a further four habitat variables were also used to help differentiate habitat types (Table 4.2). The previously-surveyed channel outlines were used as templates for all the hand-drawn habitat maps. A tape measure was laid between the marked cross-sections, to guide mapping of the distributions of different sized substrata. An overlay of the flow types was than drawn. Because flow types change with discharge, both were recorded on the same day. The maps were scanned in and imported to ARCVIEW 2.3a for digitising and the calculation of the proportions of different habitat variables. The digitised flow and substratum maps for Sites 1, 2 and 3 are presented in Chapter 5.

Measuring discharge

At each site, at the time of habitat mapping, measurements for discharge calculations were taken at three different cross-sections. For each site, mean column velocity and water depth were measured at eight to

15 points across each of the three cross-sections. Velocity was measured using a Marsh McBirney Electronic Flo-Mate current meter on a top-setting wading rod. The top-setting wading rod automatically sets the height of the current meter to the "six-tenths depth", which occurs at four-tenths of the depth (0.4D) up from the channel bed (Gordon *et al.* 1992). This is where the mean current velocity of a water column is theorised to occur. Aggregate discharge was calculated using the velocity-area method described by Gordon *et al.* (1992). The discharge values calculated for each of the three cross-sections at each site were averaged to provide a single value for each site on the day. Further discharge data, and a summary of patterns of surface flow and impacts thereon are provided as Appendix 1.

Table 4.2 **Habitat descriptor codes.** Codes simplified from King and Schael (2001). Substrata classified according to the Wentworth scale (Rowntree & Wadeson 1999).

Code	Substratum Type	Code	Flow Type	Code	Other
SI	Silt	**ST**	still	**MV**	marginal vegetation
SA	Sand	**BF**	barely flowing	**IV**	in-channel vegetation
GR	Gravel	**SS**	slow smooth	**OR**	organic litter
CO	Cobble	**MR**	medium rippled	**AL**	Algae
BO	Boulder	**TR**	trickle		
BR	Bedrock	**CA**	cascade		

4.2 Biological characteristics

The riparian and hill slope vegetation and the aquatic macro-invertebrates were monitored over the two successive summer periods after the fires. Few such data sets exist in South Africa, the only other known one being Delny Britton's Ph.D (Britton 1991) who monitored recovery of a Western Cape river for two years following a control burn. She monitored the physico-chemical characteristics of the river water and the response of the aquatic macroinvertebrate fauna but did not investigate the riparian flora in any detail.

Vegetation belt transects

Three belt transects were established at each of the three sites to record the distribution, density and condition of all plant species. These transects were positioned along the surveyed cross-sections and were numbered by site and in sequence within a site from upstream to downstream, as done for the cross-sections. At Site 2, as there were five cross-sections, the vegetation transects were positioned on cross-sections one, three and five. A belt transect consisted of back-to-back, metre-square, groundcover plots along the complete length of each transect. A total of 468 one-metre squared plots were sampled each year (Table 4.3).

Each plant in each plot was identified to species, and the areal coverage of each species (judged by eye) in each plot was recorded as a percentage, together with the height of the tallest representative of each species present and the condition of the plant (estimated as dead, struggling or healthy). A specimen of each species was collected from outside the plots, and pressed for later identification by plant taxonomists at the Compton Herbarium in the Botany Department at the University of Cape Town (Appendix 2). Five measurements of soil penetrability, to a maximum depth of 1 metre, were made in each metre-square plot by hammering a rod into the ground inside each corner and in the middle. All vegetation data were entered into the vegetation database TURBOVEG V1.99h (Hennekens 1996). TURBOVEG outputs data files that are compatible with most common analysis packages such as Microsoft EXCEL, CANOCO (ter Braak and Smilauer 1998) and PRIMER (Clarke and Gorley 2001).

Table 4.3 **Vegetation transects sampled at the study sites and the number of metre squared plots sampled along each.** The transect number is the same as the cross-section on which it was positioned. The number of plots sampled along each cross-section indicates the length of that cross-section.

Site	Transect number	Transect length (m, # plots)
1	1-1	61
	1-2	60
	1-3	51
2	2-1	49
	2-3	78
	2-5	98
3	3-1	22
	3-2	24
	3-3	25
Total	**9**	**468**

Macroinvertebrates

The standard method for water quality monitoring in use at the time of sampling was SASS4 (Thirion *et al.* 1995). SASS4 is a macroinvertebrate monitoring tool that assesses water-quality impacts and has been used nationally on South African rivers for the purposes of biomonitoring. This has subsequently been upgraded to SASS5 (Dickens and Graham 2002) and is described in section 2.4. The SASS4 methodology was not used as a more comprehensive sampling strategy, after King and Schael (2001), that separated the two components of aquatic biotopes (flow and substratum type, see Table 4.2) was chosen.

This allowed for more samples to be collected at a finer resolution than could have been done using SASS4 and allowed abundance data to be generated based upon laboratory identifications of the organisms collected.

A total of 47 invertebrate samples was collected during each summer sampling period i.e. 94 in total. Habitat descriptor codes (Table 4.2) were used to locate areas for the collection of the invertebrate samples. The combination of flow and substratum types together describe the sampled biotope. At each site all available combinations of the six substratum and six flow types were identified on the habitat maps (see Section 4.1), and at least one sample taken from each recognised combination (biotope). The digitised maps of the aquatic maps were used to guide the area sampled for each biotope. The biotopes sampled are summarised in Table 4.4; their areas were calculated from the maps. Specimens were collected using a rectangular hand net 0.20 m x 0.15 m with a 250 μm mesh size. A soft plastic hand brush was used to scour bedrock and to reach between boulders. Gravels were disturbed by hand, whilst silt and sand were collected and later searched for animals through flotation and sieving. Specimens were preserved in 70% alcohol and identified to family level in the laboratory using SASS taxonomic guidelines (Thirion *et al.* 1995; Gerber and Gabriel 2002). Abundances were reduced to a total number of individuals per m^2 by dividing the total number of individuals from each sampled biotope into the area sampled (Appendix 3).

4.3 Statistical analyses

Vegetation data

Site 3 was exluded from this analysis as it differed to Sites 1 and 2. Sites 1 and 2 both were principally invaded by one exotic tree species. Site 3, on the other hand, had both non-woody (i.e. grass) and woody exotic species present in the post-fire community. These were *Pennisetum clandestinum* and *Acacia* spp. The effects of the woody and non-woody components could not be separated out during the analyses. For sites 1 and 2, species cover data were 4th root transformed to achieve a balance between the influences of common and rare species. The transformation boosts the contribution that a species with low areal cover or abundance makes towards the community composition. This was done so that fine adjustments between the two different community years were more easily detected. The cover data were then converted to a matrix of similarity between each pair of sites, using the similarity coefficient of Bray-Curtis.

Table 4.4 **The number of invertebrate samples per site in 2001 and 2002, and the kind and area of substratum-flow combination sampled.** Acronyms as per Table 4.2. Data collected in April 2001 and March 2002.

	2001			2002		
Site	**Sample #**	**Habitat description**	**Size (m)**	**Sample #**	**Habitat description**	**Size (m)**
1	1	TR on CO & GR	0.8 x 0.2	1	ST with MV on CO & SA	1.0 x 1.0
	2	BF with OR on SA & GR	0.5 x 0.5	2	ST on CO & SA	2.0 x 1.0
	3	SS on CO, GR & SI	2.0 x 0.5	3	ST on SA	2.0 x 1.0
	4	BF with MV on SA & GR	3.0 x 0.5	4	ST on BR	1.0 x 0.5
	5	MR on SA	1.0 x 0.5			
	6	MR on SA & CO	2.0 x 1.0			
2	1	SS on BO, GR & SA	2.0 x 0.5	1	BF on SA	1.0 x 1.0
	2	BF on SA & SI	2.0 x 5.0	2	ST with MV on SA	1.0 x 0.5
	3	MR with MV	0.8 x 5.0	3	MR with OR on SA	1.0 x 0.5
	4	MR on BO & SA	2.0 x 1.0	4	MR on BR	1.0 x 0.3
	5	SS with MV on BO & SA	3.0 x 0.5	5	CA with AL on BR	0.3 x 0.3
	6	MR on BR & SA	2.0 x 0.5	6	ST on BR & SA	2.0 x 2.0
	7	BF on SA & BR	2.0 x 3.0	7	BF on CO & SA	1.0 x 0.5
	8	MR on SA & GR	1.0 x 0.5	8	SS on SA	0.5 x 0.5
	9	MR on CO	1.0 x 0.5	9	SS on BO & SA	1.0 x 0.5
	10	MR on BO & GR	4.0 x 1.5	10	MR on CO, BO & GR	2.0 x 0.3
	11	MR with MV on SA & CO	5.0 x 1.5	11	CA on BO & CO	1.0 x 0.3
	12	BF with MV on SA	1.0 x 1.5			
3	1	BF with MV	1.5 x 0.2	1	MR on CO, BO & GR	1.5 x 1.0
	2	BF with OR on SA, GR & BR	0.5 x 0.2	2	BF on BO & SA	1.0 x 0.5
	3	CA with IV on CO & GR	1.0 x 0.2	3	BF with MV	2.0 x 0.5
	4	CA on BO	0.2 x 0.2	4	CA on BO	1.0 x 0.5
	5	BF on SA	2.0 x 3.0	5	BF on SA & SI	1.0 x 1.0
	6	BF on BO	1.0 x 3.0	6	ST on GR & SA	0.5 x 0.5
	7	MR on BR	5.0 x 2.0	7	MR on BR	2.0 x 0.3

Non-metric Multi-Dimensional Scaling (MDS) techniques were used to display the biotic relationships between the three transects at each site and between samples from each site individually. PRIMER (Clarke and Warwick 2001) was employed to cluster the site samples and produce MDS ordination scatter diagrams and CLUSTER dendrograms of the relationships between site samples. Using these, the riparian zone was identified as sets of contiguous quadrats from the water's edge and up the bank on every transect, which consistently grouped together each year. These quadrats fell within the marked floodline on the surveyed cross-sections, validating delineation of the riparian community. The same riparian quadrats on each transect were selected for each year.

For each site the sections of the three belt transects identified as riparian were combined. Thus for each site there was one riparian data set in each year and comparisons were made between the two years. The total number of both native and alien species was recorded, and the mean richness within each plot calculated. Measures of Shannon-Weiner species diversity and Simpson's equitability were also calculated for each site (Zar 1996). In order to identify the dominant species at each site, species-cover data were ranked by frequency of occurrence (100% = a species occurred in all sample quadrats at a site) and total cover (the average over all sample plots within a site), these two measures capturing different aspects of species dominance. Each species was classified into one of the six growth forms (Table 4.5) according to Goldblatt and Manning (2000), and the number of species belonging to each group and their total cover was summed.

Table 4.5 The six growth forms used to classify species groups, with heights referring to the adult of the species. Growth form nomenclature follows Goldblatt and Manning (2000).

Growth form	Definition
1 forb	A broad leafed herbaceous plant other than graminoids.
2 graminoid	Plants in the family Juncaceae, Cyperaceae, Poaceae and Restionaceae.
3 ericoid	Shrubs with ericaceous leaves.
4 shrub	A low or medium sized woody perennial plant often with multiple stems (<1m).
5 small tree	A large woody perennial plant usually with multiple stems or with main trunk (2-10m).
6 tree	A tall woody plant with main trunk, branches and a distinct elevated crown (>10m).

The maximum height of each species occurring within each plot was measured to inform on the vegetation structure at each site. For consistency, the methods described by Galatowitsch and Richardson (2005) were used to categorise plant height into four vertical strata (m): <0.5, 0.5-2.0m, 2.0-5.0 and >5.0m. For each quadrat, each species was placed into one of the four height categories, and the cover of that species within the quadrat was assigned to the respective strata. The total plant cover occurring in each height class was calculated for each site. The tallest stratum with more than 25% of the total vegetation cover was assigned as the dominant height class, and the number of occupied strata and relative cover in each was used to calculate a simple measure of Plant Height Diversity using the Shannon-Weiner diversity index. Although not all of the plant's foliage may fall within the assigned stratum, for the purposes of this thesis this limitation was accepted.

Invasion of alien plant species into the Silvermine catchment was so extensive that few, if any, areas were suitable to serve as appropriate reference sites against which recovery could be assessed. Acknowledging the difficulties in assigning appropriate restoration targets, the recovery of the riparian vegetation was assessed by comparing the species composition of each site two years post fire to a reference condition database for Riparian Scrub from mountain stream and foothill reaches of Western Cape rivers (Reinecke *et al.* 2007).

A further *post hoc* analysis investigated the effects of the follow-up clearings by Working for Water at site 2 on the proliferation of weedy perennial grasses, specifically *Ehrharta setacea*. The regenerating *Acacia* spp. were removed between the 2001 and 2002 survey, and so the total percentage cover of *Acacia* spp. in each plot during these years was calculated along with that for *E. setacea,* and compared. The percentage data were arcsine transformed to improve model fit and behaviour. The relationship between the changes in the cover of these species was analysed using ordinary least-squares regression implemented in the R programming environment (R Development Core Team 2004).

Aquatic macroinvertebrates

The distribution of aquatic macroinvertebrates amongst the biotopes available was investigated using a range of multivariate statistics rather than univariate methods, since univariate methods collapse data into single measures where much of the community structure and complexity is lost; multivariate methods capture rather than reduce this complexity (Field *et al.* 1982; Clarke and Warwick 2001). The same Non-Metric multidimensional scaling techniques in PRIMER, as used for the initial analyses of the vegetation data, were used again here to ordinate the samples into affiliated groupings based upon familial community composition. In order to determine whether any of the habitat descriptor codes (Table 4.2, flow and substratum types) and/or the year in which the sample was collected were responsible for the groupings formed, ANOSIM (analysis of similarities, Clarke and Warwick 2001), a non-parametric permutation procedure that is analogous to ANOVA (analysis of variance), was employed. The aim was to use the habitat descriptor codes and associated habitat maps, which recorded changes in available habitat between the two sampling periods, to search for justifiable changes in community composition that could be attributed to changes in the available habitat.

The ANOSIM statistic tests for differences in similarity between sites and compares these to differences among replicates within sites. R, the test statistic, can only lie somewhere between -1 and 1. At the uppermost level, R equal to 1 indicates that all replicates within sites are more similar to each other than to any other replicates from other sites. The null hypothesis was that there were no significant differences in community composition between the three sites. R is approximately zero if the null is true. R usually falls between 1 and 0 indicating some degree of discrimination between sites; $R<0$ would indicate similarities across sites were higher than those within sites. The R value is more important than the significance level (which is based upon the number of replicates, and may be insignificant as a result of insufficient replicates) as it gives an absolute measure of how separated the groups are. $R>0.75$ may be interpreted as groupings being well separated, $0.75>R>0.5$ as groups that are clearly different but overlapping and 0.25>R as groups that are barely separable. The global R searches for differences and if the null is rejected then it is useful to assess single pairs of comparisons for similarities.

The habitat descriptor codes chosen to be representative of invertebrate biotopes were reduced *a posteriori* into coarser habitat units to assess whether simpler, more meaningful groupings could be distinguished. The flow types were reduced to two categories only: fast and slow. The substratum types were reduced to two different combinations. The first set of categories investigated was rock (bedrock, boulder and cobble), sand (sand and silt), rock/sand, sand/vegetation (marginal), rock/sand/vegetation, rock/vegetation and vegetation. The second set of categories was bedrock, coarse (boulder and cobble), mixed (boulder, cobble, gravel, sand or any combination thereof), fine (gravel, sand and silt) and marginal vegetation. Where differences were shown to exist, these were tested further, using the SIMPER 'similarity percentages' routine (Clarke and Warwick 2001).

SIMPER determines how individual taxa contribute towards the separation of two groups of samples, or the closeness of samples within a group. The results from this analysis present the percentage contribution of each taxon towards the sample identity; the taxa are listed in decreasing order of such contribution. The average dissimilarity (or similarity) between two pairs of samples is represented as the sum of the individual contributions made by different taxa. The descending order in which the taxa are listed and their relative contributions towards this total average dissimilarity is the first important factor in determining which taxon may be responsible for group separation. The second element associated with this is the ratio between a taxon's average contribution and the standard deviation (SD) of that contribution across all pairs of samples making up this average. A good discriminating taxon is one with a high average abundance that contributes relatively consistently for all pairs of comparisons made, which thus has a low SD and a higher ratio >1. In the same way, yet perhaps of less practical significance, one can examine the contribution that each species makes to the average similarity within a group. The more abundant a species is within a group (by numbers of individuals), the more it will contribute to the intra-group similarities. It typifies the group if it is found consistently in each sample constituting the group. This, however, says nothing about whether the species is a good discriminator of one group from another; it may be very typical of a number of groups.

In order to identify the dominant families at each site, family level abundance data were ranked by frequency of occurrence (100% = a family occurred in all sample quadrats at a site) and total abundance (the average over all samples at a site), these two measures capturing different aspects of family-level dominance. Frequency distributions of the families present were calculated in two different ways: first on the basis of abundance (numbers of organism present in each sample) and then by presence/absence, which reduces the effect of an over-dominant organism.

These data were compared to an aquatic macroinvertebrate reference condition for upland (mountain stream and foothill) zones of Western Cape rivers (Dallas and Day 2007). The database contains a list of expected taxa that includes details pertaining to seasonality and biotope preferences. This database was valuable as it takes into account the intrinsic variability amongst reference conditions. Ranges are set for five categories, from 'richer than reference', a biological hotspot, down to 'well below reference' and 'impoverished'.

Since the strict protocol of the SASS5 (Dickens and Graham 2002) methodology was not followed it was not possible to make direct comparisons to other data sets from the Silvermine River (Fowler and Harding 2000; Snaddon 2007) or to other SASS reference data bases (Dallas *et al.* 1998; Dallas and Ratcliffe 2004; Ollis *et al.* 2006; Dallas and Day 2007; Snaddon 2007). Nonetheless, since SASS (Thirioin *et al.* 1995) is based upon the sensitivities of the organisms to pollution and habitat degradation, it was possible to use these sensitivity ratings, inherent in the method, to generate pseudo-SASS scores. These were calculated for each year and compared inter-annually. Two different analyses were performed using the pseudo-SASS scores. First, samples were chosen to represent pseudo-SASS biotopes (Table 4.6) in order to calculate the pseudo-SASS scores. The sensitivity ratings were also used to generate the proportions of tolerant and sensitive organisms present. Using the SASS sensitivity scores (Thirion *et al.* 1995) for taxon groups, those organisms with scores from 1-7 were ranked 'tolerant', while those with scores from 8-15 were ranked 'sensitive'.

An assessment of Present Ecological Status (PES) was used to class the overall potential for degradation, through anthropogenic disturbances, at the three study sites. These disturbances "include abiotic factors, such as water abstraction, weirs, dams, pollution and dumping of rubble, and biotic factors such as the presence of alien plants and animals" (Dallas 2000). Each assessed category is given a score and each category contributes a weighted percentage toward a total score. PES scores for the Silvermine River were calculated following the procedures required for assessing Habitat Integrity Status (Dallas 2000). The method generates in-stream and riparian status scores separately and was applied according to the guidelines prepared by Brown *et al.* (2001), which were developed to assess PES in the absence of hydrological records for rivers in the Western Cape. The calculated scores are used to classify the assessed river reaches as A to F class PES according to percentages described in Table 4.7.

Table 4.6 **Macro-invertebrate samples chosen to represent pseudo-SASS samples for the comparative analysis between sample years.** Habitat descriptor codes as per Table 4.2. SASS codes: SIC = stones in current; SOOC = stones out of current; MV = marginal vegetation; and SA = Sand/Silt.

	2001			2002		
Site	Sample #	Habitat description	SASS code	Sample #	Habitat description	SASS code
1	1	TR on CO & GR	SIC	-	-	No SIC
	2	BF with OR on SA & GR	SOOC	2	ST on CO & SA	SOOC
	4	BF with MV on SA and GR	MV	1	ST with MV on CO & SA	MV
	5	MR on SA	SA	3	ST on SA	SA
2	10	MR on BO & GR	SIC	10	MR on CO,BO & GR	SIC
	1	SS on BO, GR & SA	SOOC	7	BF on CO & SA	SOOC
	3	MR with MV	MV	2	ST with MV on SA	MV
	2	BF on SA & SI	SA	8	SS on SA	SA
3	4	CA on BO	SIC	1	MR on Co, BO & GR	SIC
	6	BF on BO	SOOC	2	BF on BO & SA	SOOC
	1	BF with MV	MV	3	BF with MV	MV
	5	BF on SA	SA	5	BF on SA & SI	SA

Table 4.7 **Preliminary present status classes** (Kleynhans 1996, cited by Dallas 2000).

Class	Description	Score (%)
A	Unmodified, natural.	
B	Largely natural with few modifications. A small change in natural habitats and biota may have taken place, but the assumption is that ecosystem functioning is essentially unchanged.	80-89
C	Moderately modified. A loss of change in natural habitat and biota has occurred, but basic ecosystem functioning appears predominately unchanged.	60-79
D	Largely modified. A loss of natural habitat and biota and a reduction in basic ecosystem functioning is assumed to have occurred.	40-59
E	Seriously modified. The loss of natural habitat, biota and ecosystem functioning is extensive.	20-39
F	Modifications have reached a critical level and there has been an almost complete loss of natural habitat and biota. In the worst cases, the basic ecosystem functioning has been destroyed.	0-19

Finally, relevant data and discussion on general management and restoration (Fowler and Harding 2000; Harding 2000) and (Reinecke and King 2003; Reinecke *et al.* 2003), and the pertinent biotope and community composition results of two B.Sc (Hons.) theses (Shelton 2003; Mann 2004) that followed this work through the third and fourth year's recovery are drawn into the thesis in chapter 7.

5 PHYSICAL SITE CHARACTERISTICS

This chapter presents the site-level physical data in the form of surveyed cross-sections and habitat maps. The cross-sections depict channel shape while the habitat maps summarise the proportions of the two biotope components, flow and substratum type. In both instances, the results compare differences between the two sampling periods, summer 2001 and summer 2002.

5.1 Channel dimensions

Study site 1, previously invaded by the tree *__Pinus pinaster__* *(Cluster Pine) prior to the 2000 fires*

Three cross-sections were established at this site, which was approximately 50 m long by 60 m wide (Figure 5.1). Since the right bank was going to be landscaped, the sampling of vegetation was focussed on the left bank as this was where spontaneous succession of vegetation was expected to occur. The left bank sloped gently (Figure 5.2) towards the river channel. Each cross-section was 50-60 m long. All the cross-sections were surveyed twice (Table 4.1, Figure 5.2). There were no obvious changes in channel shape over the year between the two summer periods. Each cross-section remained the same shape. Since a vegetation plot was placed along every metre of each cross-section, there were approximately 50 and 60 groundcover vegetation plots sampled along each of these cross-sections. Two different discharge (Q) measurements are shown on the cross-sections. The solid line shows the water level at winter low flow ($Q = 0.02\ m^3s^{-1}$). The dotted line shows the highest stage where flood debris was deposited after a large flood during the winter of 2001. The macro-channel was approximately 10 m in width, the active channel 5 m in width and with a water depth of 0.2 m at summer low-flow. Further measurements of discharge are given for all study sites in Appendix 1.

Study site 2, previously invaded by the tree *__Acacia saligna__* *(Port Jackson willow) prior to the 2000 fires*

This was the largest site, being 100 m long and 80 m wide (Figure 5.3A). Five cross-sections were established, each between 50 and 100 m in length. Thus there were between 50 and 100 vegetation groundcover plots sampled along each cross-section. Each cross-section was surveyed twice (Table 4.1). Site 2 was chosen because of the high degree of mobility of the channel, with high, almost vertical, sand banks edging the river (Figure 5.4). Only the macro-channel portions of each of the five cross-sections are shown. The cross-sections at this site extended a further 20-30m up the left bank into the terrestrial vegetation and these portions are not shown. The channel shapes of cross-sections 2-1, 2-2, 2-3 and 2-5 remained the same between the two sampling periods. A large portion of cross-section 2-4 was eroded during a winter high flow event during July/August 2001. A meander cut-off occurred forming a new low flow channel (Figure 5.3B). The left bank of the river at this site is uncharacteristically high, steep and sandy when compared to the other river bank profiles such as those surveyed at Site 3. The origin of this steep sandy bank is not known but was investigated by Akunji (2002); his findings are discussed in section 7.3.2. Two different discharge (Q) measurements are shown on the cross-sections. The solid line shows the water level at winter low flow ($Q = 0.12\ m^3s^{-1}$). The dotted line shows the highest stage where flood debris was deposited after a large flood during the winter of 2001. The macro-channel was approximately 25 m in width, the active channel approximately 3 m in width and 0.4 m at summer low-flow.

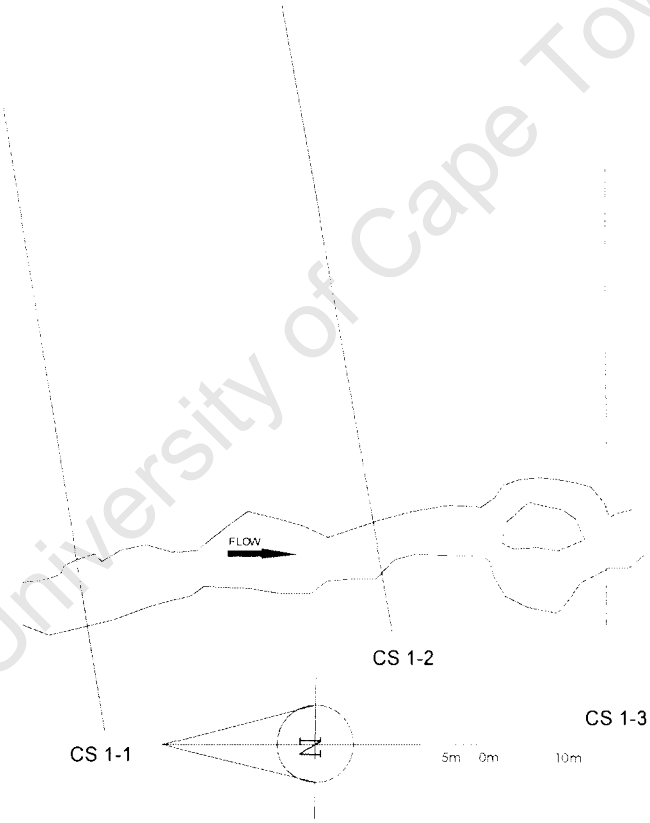


Figure 5.1 **Plan map of site 1.** Three cross-sections, each approximately 50m in length, were established. The cross-sections were numbered 1 through 3 from the upstream end. Scale 1:500.

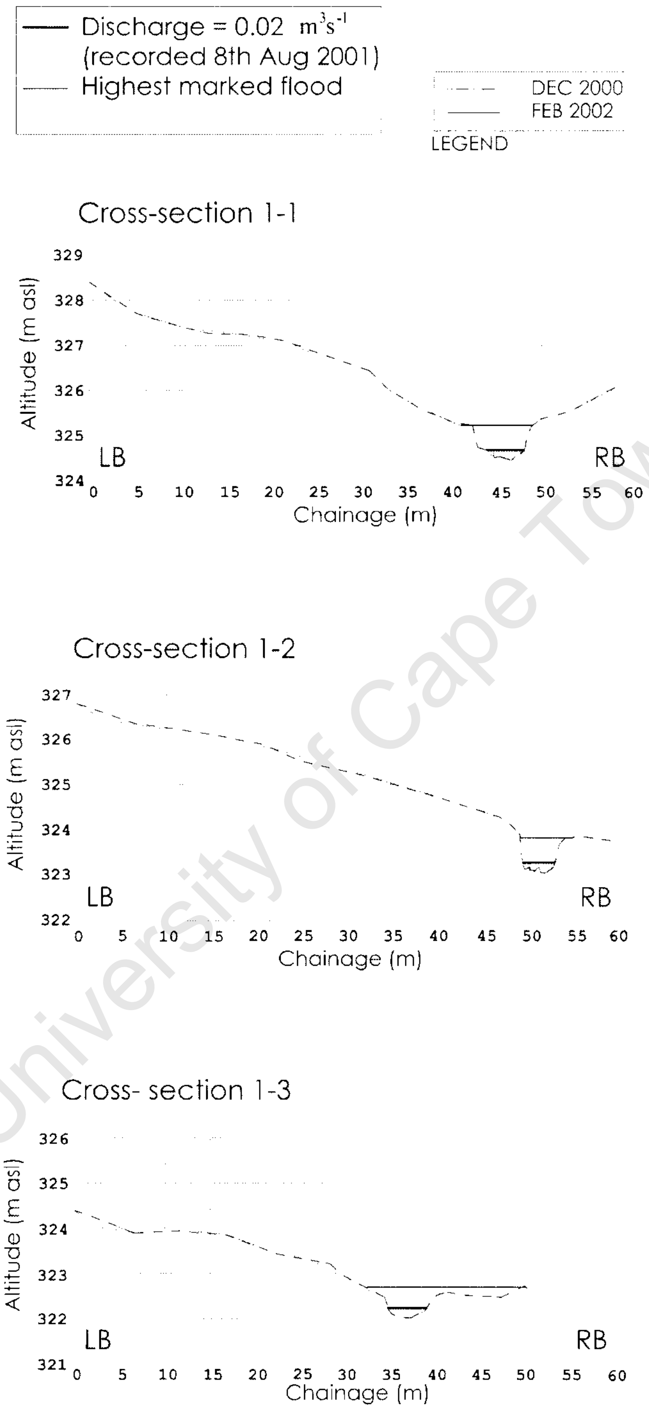


Figure 5.2 **Cross-sections surveyed at Site 1.** The study site can be located on Figure 3.2. The channel shape over two successive years is depicted. The winter low flow discharge is represented along with the elevation of the largest flood during the winter of 2001. LB = left bank, RB = right bank.

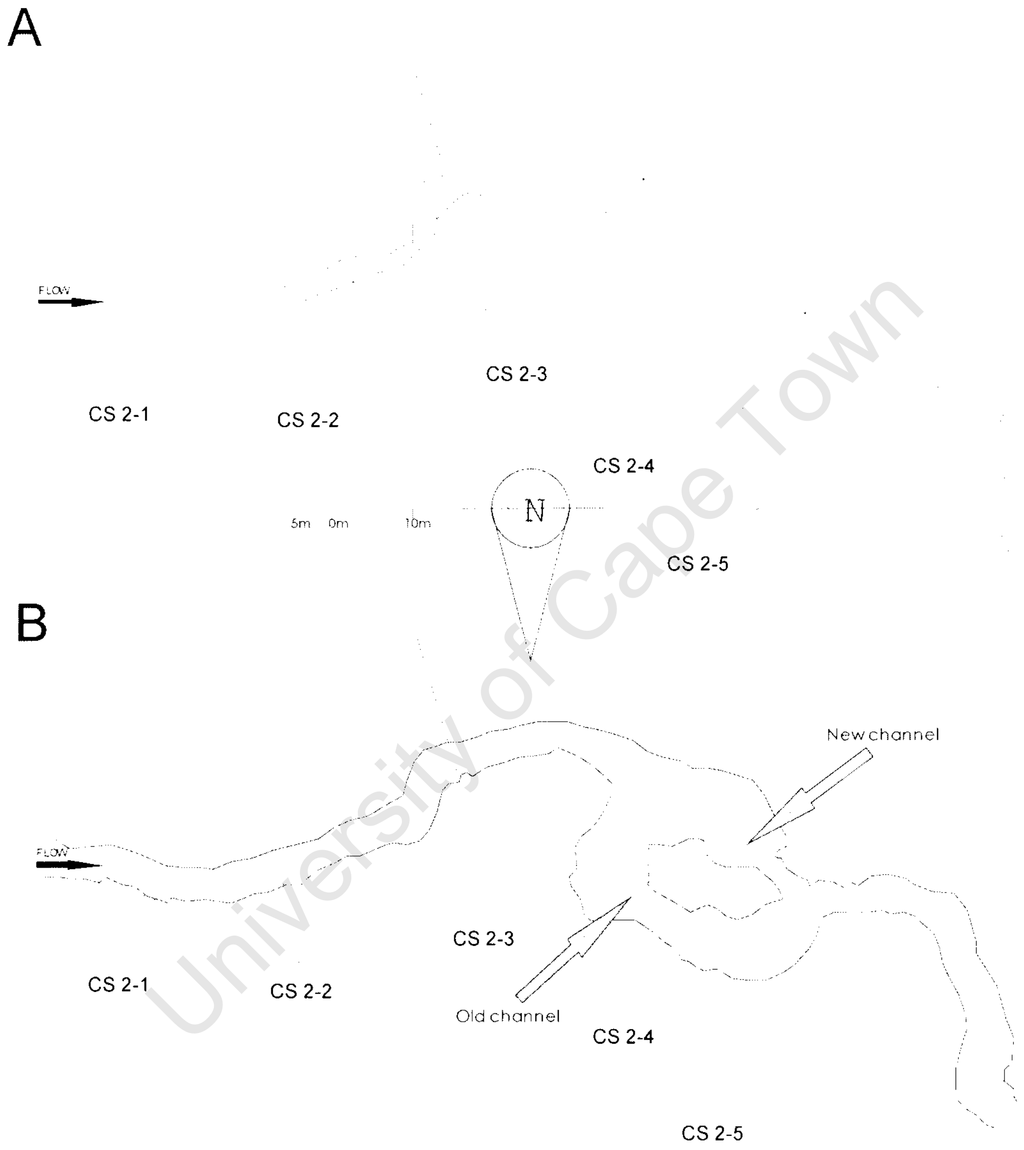


Figure 5.3 **Plan map of site 2.** Five cross-sections, each at least 50m in length, were established. The cross-sections were numbered 1 through 5 from the upstream end. Scale 1:500. The channel shape in the first year, before the winter floods of 2001, is shown in A. The channel shape after the winter floods of 2001 is shown in B.

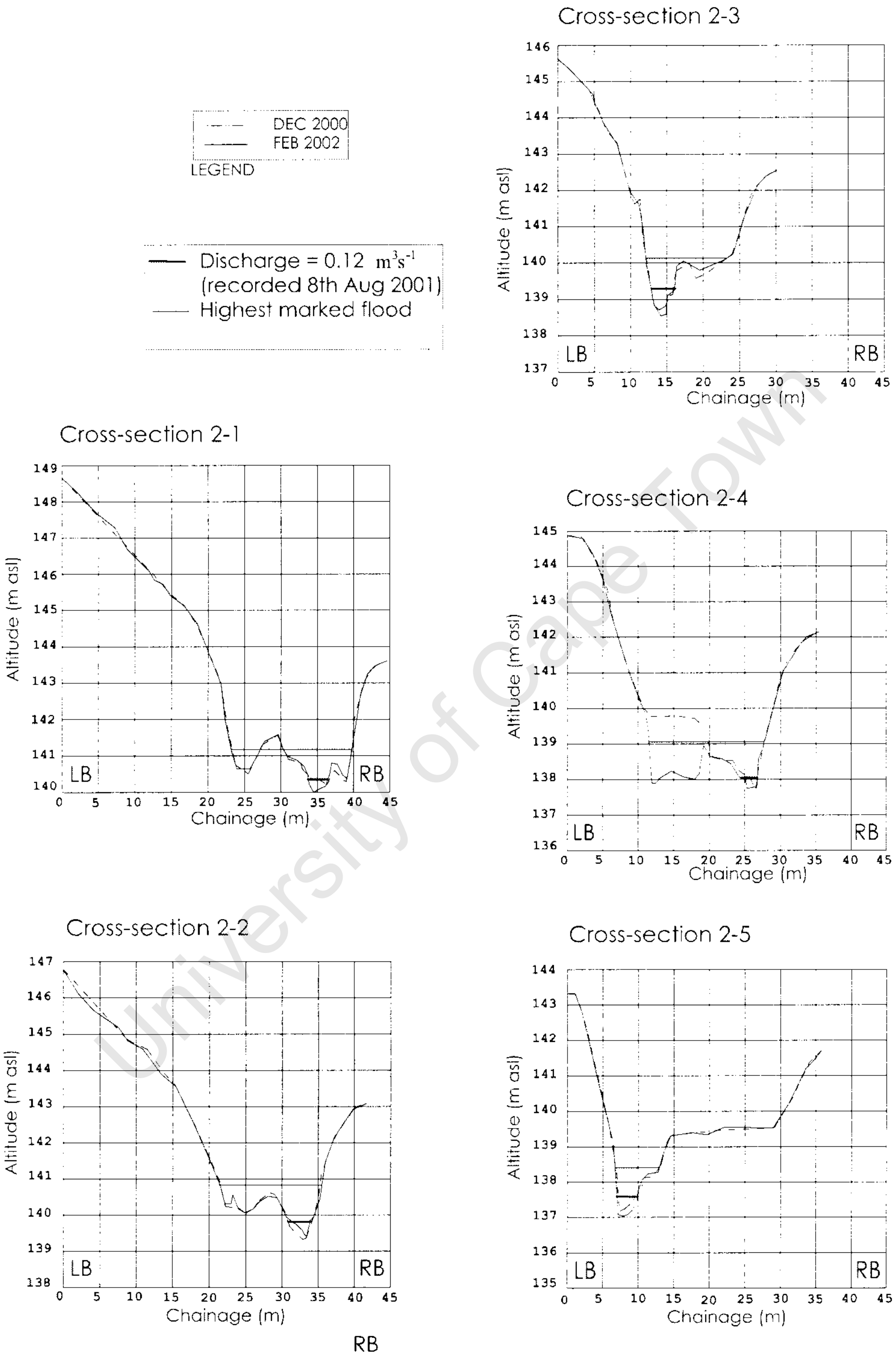


Figure 5.4 **Cross sections surveyed at Site 2.** The study site can be located on Figure 3.2. The channel shape over two successive years is depicted. The winter low flow discharge is represented along with the elevation of the largest flood during the winter of 2001. LB = left bank, RB = right bank.

Study site 3, previously a terraced agricultural area, now covered in the grass *Pennisetum clandestinum* *(Kikuyu)*

Three cross-sections were established (Figure 5.5), each approximately 25 m in length totalling around 25 vegetation groundcover plots along each. Each cross-section (Figure 5.6) was surveyed twice (Table 4.1). The channel shapes of all the cross-sections remained the same between the two sampling periods. Two different discharge (Q) measurements are shown on the cross-sections. The solid line shows the water level at winter low flow (Q = 0.15 m^3s^{-1}). The dotted line shows the highest stage where flood debris was deposited after a large flood during the winter of 2001. The macro-channel was approximately 10 m in width, the active channel 3 m in width and 0.2 m in depth at summer low-flow.

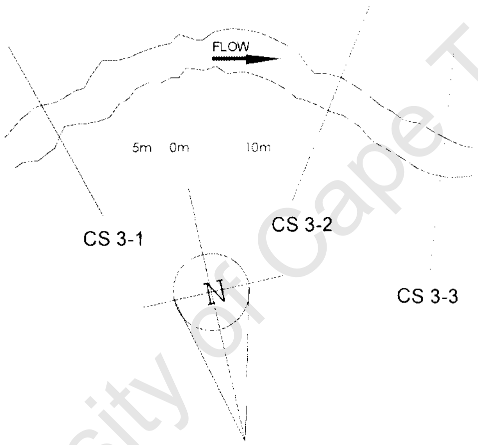


Figure 5.5 **Plan map of site 3.** Three cross-sections, each approximately 20m in length, were established. The cross-sections were numbered 1 through 3 from the upstream end. Scale 1:500.

5.2 Aquatic habitat (biotopes)

The aquatic habitat, as measured by mapping the areas of each of the 16 habitat indicators (Table 4.2) using flow and substratum type maps, differed from site to site and from year to year at each site. The flow and substratum type maps are shown in Figures 5.7-12. The percentages, by area, of each of the habitat indicators were compared to two reference sites for each of the longitudinal zones, the mountain stream (Site 1) and the foothill (Sites 2 and 3). Data for comparable habitat indicators were taken from King and Schael (2001). This was necessary since no reference sites were available on the Silvermine River. Nursery Ravine and the Disa River both on the Cape Peninsula were used as mountain stream reference sites. No foothill sites were sampled by King and Schael (2001) on the Cape Peninsula so the next two nearest in distance from the Peninsula were used. These were the DuToits River, which flows into the Theewaterskloof reservoir alongside Villiersdorp, and the Eerste River, which flows through Stellenbosch.

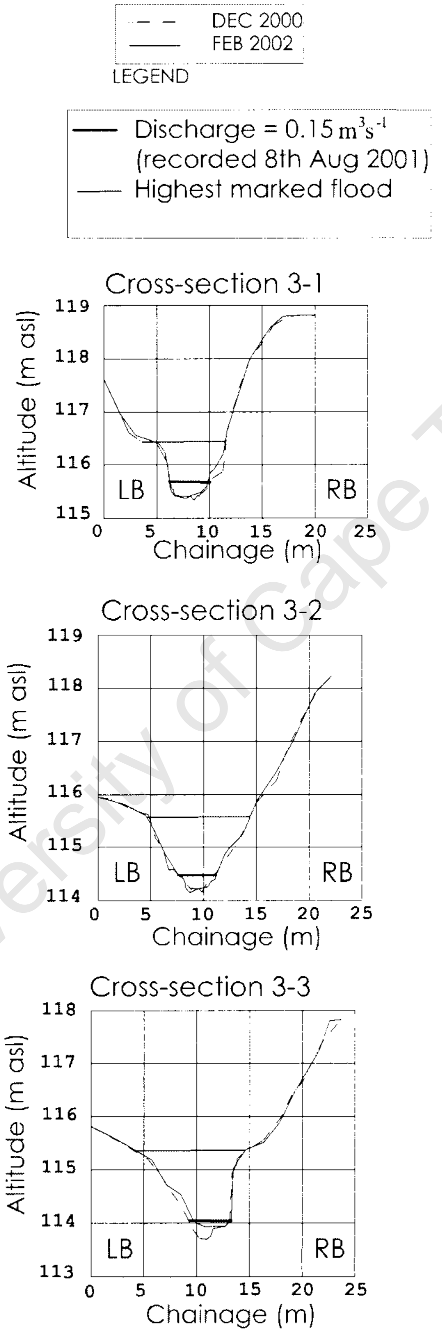


Figure 5.6 **Cross sections surveyed at Site 3.** The study site can be located on Figure 3.2. The channel shape over two successive years is depicted. The winter low flow discharge is represented along with the elevation of the largest flood during the winter of 2001. LB = left bank, RB = right bank.

Study site 1

Site 1 occurs in the mountain stream zone of the Silvermine River. In the first year (2001) one third of the substratum found in the active channel was sand, which together with gravel and silt accounted for 50% of the total present (Table 5.1). In the second year (2002) the dominant substratum types were bedrock and boulder, each constituting approximately one third of the total, and cobble constituting one fifth of the total (Figure 5.7). Combined, the total percentage of coarse particles present was 82%. Thus, from 2001 to 2002 there was a shift in the proportions of the different substratum types: a reduction in the percentages of finer substratum types (gravel, sand and silt) and an increase in the percentages of larger substratum types (bedrock and boulder).

When these percentages were compared to those recorded in the mountain stream zones of Nursery Ravine and the Disa River, the percentage of larger substrata in the first year was lower than the reference condition but was the same in the second year. This is a good indication that this component of the aquatic habitat (biotopes) has recovered back toward what is expected of a mountain stream.

Table 5.1 **The percentages, by area, of the substratum types at the Silvermine River study sites and two sets of reference condition data.** The reference rivers for Site 1 were Nursery Ravine in Newlands and the Disa River in Hout Bay and for Sites 2 and 3 were the DuToits River and the Eerste River (King and Schael 2001).

Substratum Type	Percentage by Area									
	Site 1		M.S.		Site 2		Site 3		M.S.tr	
	2001	2002	Nur.	Disa	2001	2002	2001	2002	DuT.	Eer.
Bedrock	19	32	0	1	3	8	8	8	0	6
Boulder	11	29	31	53	23	26	30	45	23	51
Cobble	20	21	59	25	15	15	20	7	71	39
Gravel	14	4	3	1	10	4	4	9	4	4
Sand	33	14	2	3	47	43	20	29	2	0
Silt	3	0	5	17	2	4	18	3	0	0
Total	100	100	100	100	100	100	100	100	100	100

Different flow types dominated 2001 and 2002 (Table 5.2). There was a complete change from slow smooth flow in 2001, when the recorded discharge through the site was 0.001 m^3s^{-1}, to virtually no flow in 2002 when the velocity was smaller than could be detected by the flow meter (< 0.001 ms^{-1}). The flow types mapped on the Silvermine River (Figure 5.8) differed widely from those for both Nursery Ravine and the Disa River mapped at a similar time of year. Slow smooth flow dominated the first year and barely flowing flow dominated the second year at the Silvermine River, both exceeding 94% of the total flow mapped (Table 5.2). The two reference rivers both had lower proportions of these two slow-flow types and higher proportions of faster-flow types, totalling 62% at Newlands Ravine and 30% at the Disa River. The dominance of low-flow at the Silvermine River was due to abstraction of water from the upper section of the Silvermine River during summer (see Appendix 1). This left little or no flow in the main channel until the larger tributaries had contributed flow into the main channel downstream.

Table 5.2 **The percentages, by area, of the flow types at the Silvermine River study sites and two set of reference condition data.** The reference rivers for Site 1 were Nursery Ravine in Newlands and the Disa River in Hout Bay and for Sites 2 and 3 were the DuToits River and the Eerste River (King and Schael 2001).

Flow Type	Percentage by Area											
	Site 1		M.S.		Site 2		Site 3		M.S.tr	
	2001	2002	New.	Disa	2001	2002	2001	2002	Dut.	Eer.
Still	0	99	4	3	0	4	0	12	1	3
Barely flowing	1	0	13	47	37	55	46	38	4	6
Slow smooth	94	0	21	20	35	18	36	30	9	13
Medium rippled	0	1	31	15	27	23	17	20	60	42
Fast turbulent	0	0	19	13	0	0	0	0	23	33
Trickle	5	0	1	0	0	0	0	0	0	2
Cascade	0	0	11	2	1	0	1	0	3	1
Total	100	100	100	100	100	100	100	100	100	100

Study sites 2 and 3

Sites 2 and 3 both occur in the foothill zone of the Silvermine River. Both showed similar trends in the proportions of substrata and flow types mapped. At Site 2 similar proportions of all substratum types were recorded during both years (Figure 5.9). Approximately a quarter of the mapped area was covered by boulders, which with the other two larger particle categories constituted 41% of the total in 2001 (Table 5.1). This increased by 8% in the second year to 49%. In both years at Site 2 smaller particle sizes (gravel, sand and silt) constituted between 51% and 59% of the substratum. When compared to the proportions of substrata found at the DuToits and the Eerste Rivers, the large amount of sand and the small proportion of cobble appear to be abnormal. The dominance of sand was not cleared by the winter floods of 2001 as more sand was deposited from the erosion of the left bank near cross-section 2-4. Upstream of cross-section 2-4, the substratum map (Figure 5.9) clearly shows a change between years with a larger proportion of boulder and cobble in the second year, and a decrease in the proportions of gravel and sand. Downstream of cross-section 2-4, where the meander cut-off occurred, there was still a larg proportion of sand present in the second year, which is unnatural. The erosion of this bank will continue if it remains unstablised and this will retard recovery of the aquatic habitats available to macroinvertebrates (see Chapter 7).

A similar situation occurred at Site 3 (Figure 5.11). From one year to the next there was a slight increase in the percentage of boulders and a corresponding decrease in the percentage of cobble. A smaller percentage of silt was found after the winter floods. When compared to the reference sites, a larger amount of sand and a much smaller amount of cobble was found at Site 3 than was present at either of the DuToits and Eerste Rivers. Fine particles constituted between 41% and 42% of the total mapped substrata at the Silvermine while there were much lower percentages on the DuToits and Eerste respectively: 6% and 4%. There was no obvious erosion taking place from the river banks at Site 3 so this excess of sand was attributed to the erosion and transport of sand from Site 2 upstream.

There were no large differences in the proportions of flow types between the two mapped years at Sites 2 and 3 (Figures 5.10 and 5.12). Both showed a dominance of slower flow types, and since the habitats were mapped during the lower flowing summer months this is not surprising. King and Schael (2001) mapped a larger proportion of faster flow types, for example medium rippled and fast turbulent flow on the two reference rivers, the DuToits and Eerste.

5.3 Summary of habitat changes between years

The river banks of all three sites on the Silvermine were generally stable over the two years. The majority of the cross-sections surveyed remained the same. This was not the case in the central section of Site 2 however, where a meander cut-off occurred at cross-section 2-4. The left bank of the river at this point is high, sandy and unstable.

The proportions of flow and substratum types mapped showed changes between sites and years. In the mountain stream (Site 1), the reduction in proportion of fine substratum types and corresponding increase in larger particle sizes from 2001 to 2002 approached proportions that resembled reference rivers. However the abstraction of water from the mountain stream reduced the proportion of faster-flowing waters that are characteristic of reference mountain streams.

In the foothills (Sites 2 and 3), erosion of the high sandy bank at Site 2 resulted in higher proportions of sand and lower proportions of cobble than would normally be found. The unstable bank here is uncharacteristic when compared to the rest of the foothill. Ongoing erosion of this river bank will continue to deposit sediment into the channel at Site 2 that will be transported downstream. The proportions of substratum types present in this foothill will not recover towards reference conditions unless this bank is stabilised. This could be part of a restoration plan that includes input from the riparian vegetation and the macroinvertebrate studies (see Chapters 6 and 7 respectively). If the bank is stabilised and erosion of the sandy bank halted, winter floods will transport the excess sediment downstream and out of the foothill. Riparian vegetation will come to occupy the newly stabilised bank and the aquatic habitats (biotopes) available in the channel will come to resemble a classic foothill, with the presence of faster flow types and larger river-bed particles and clearer physical definition of the difference classes of substrata. This should have positive effects on boosting the diversity of macroinvertebrates in the river.

Figure 5.7 **Substratum maps for Site 1.**

Figure 5.8 Flow maps for Site 1.

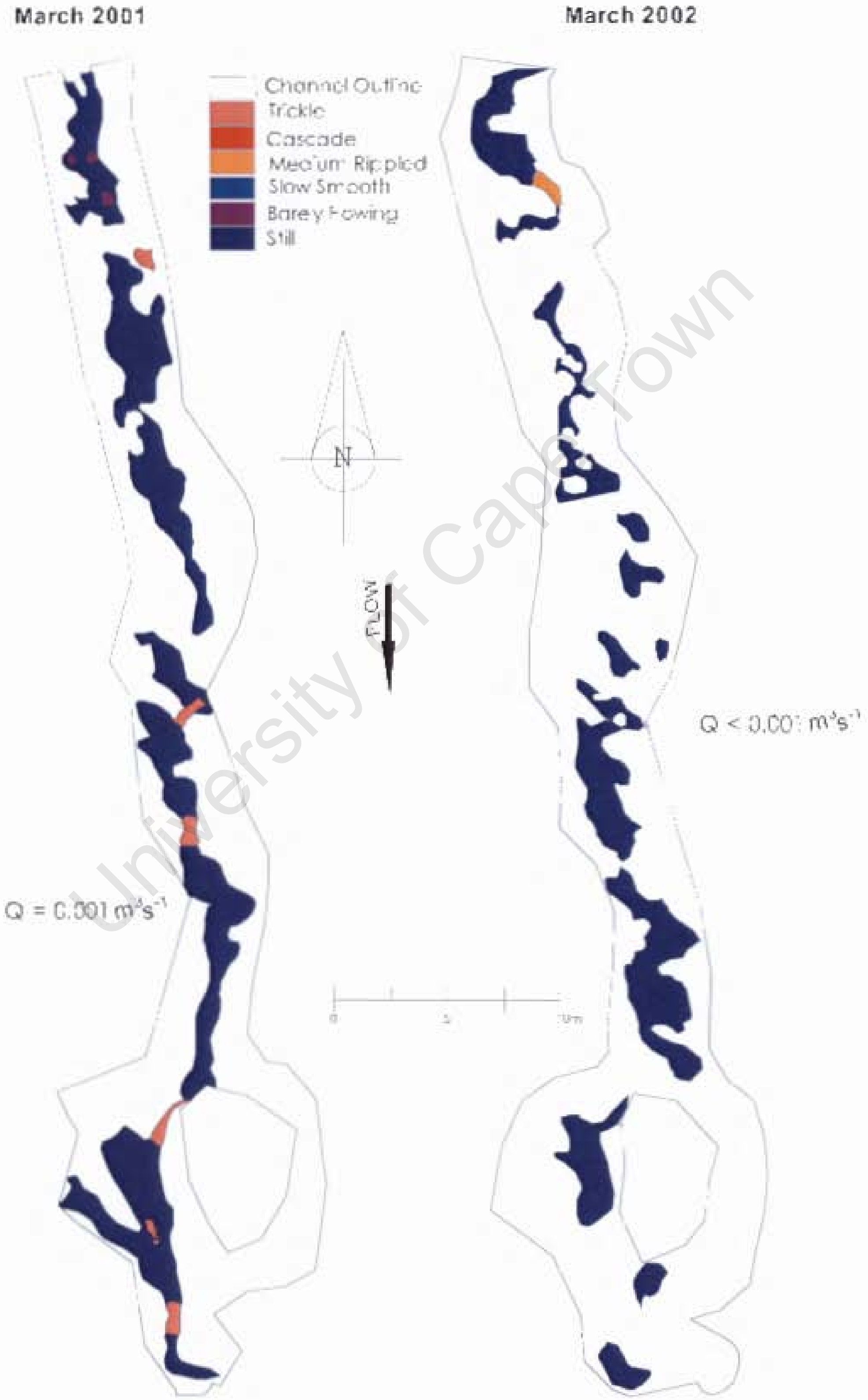


Figure 5.9 Substratum maps for Site 2.


March 2001

March 2002

Figure 5.10 Flow maps for Site 2.

Figure 5.11 Substratum maps for Site 3.

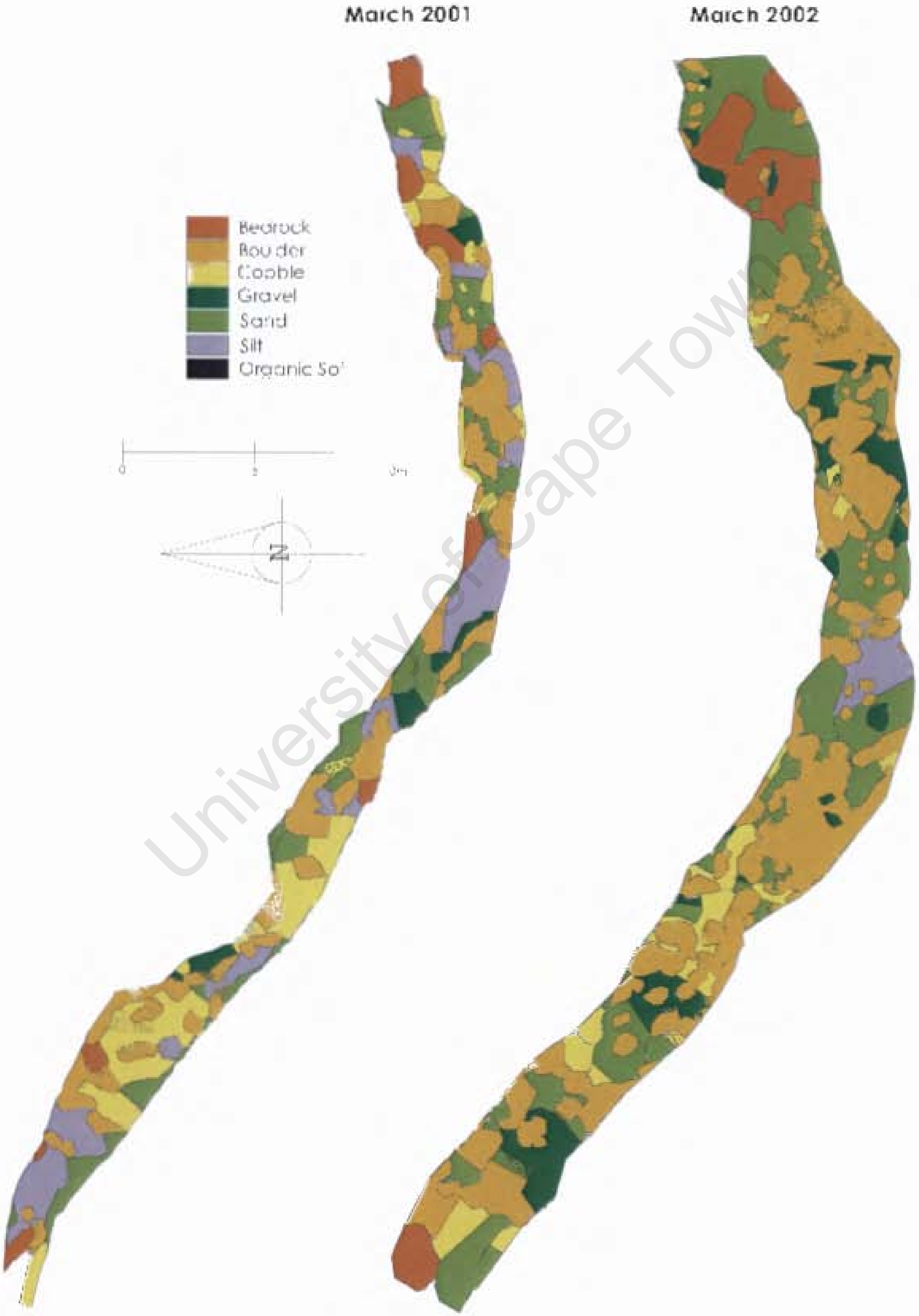
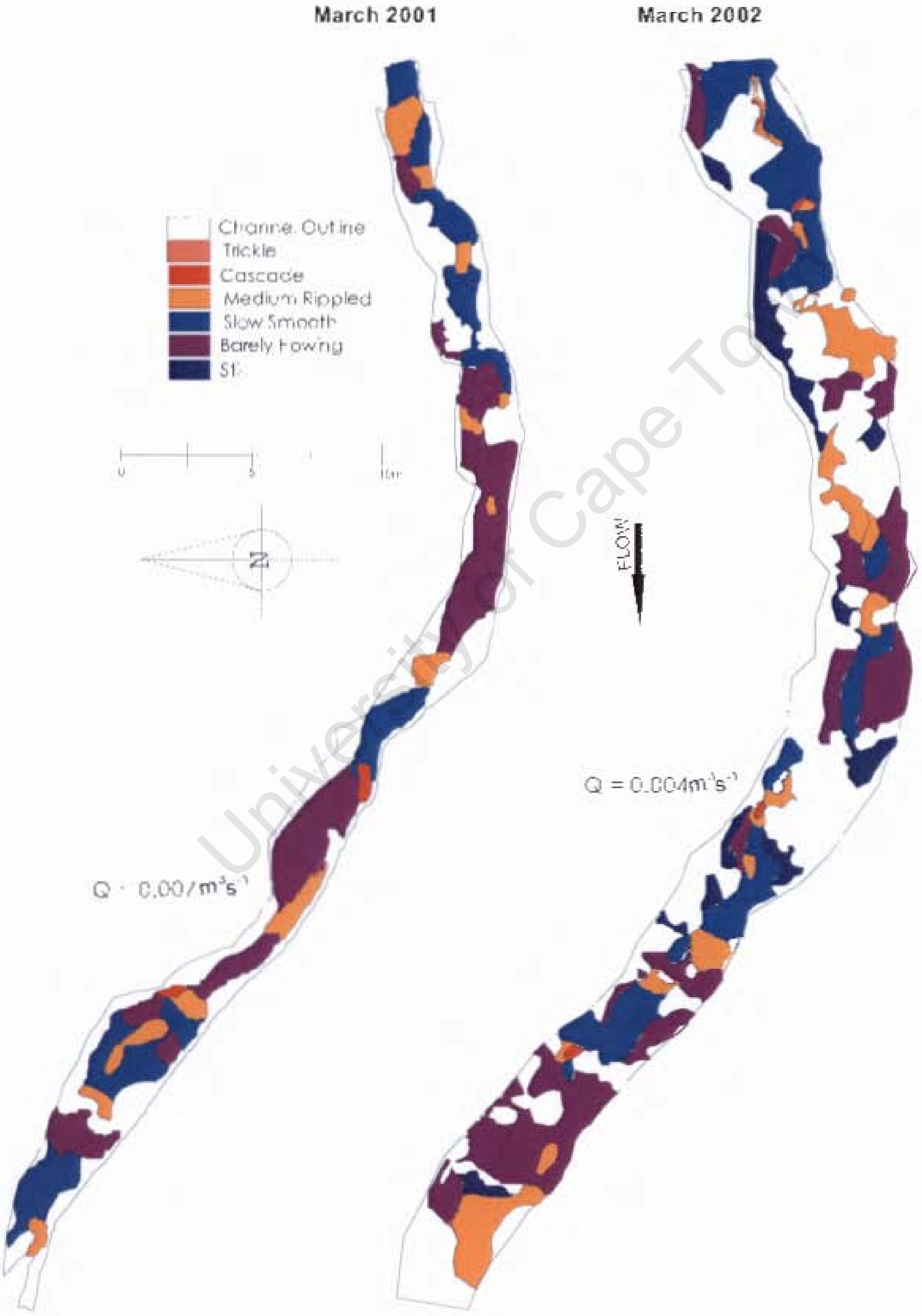


Figure 5.12 Flow maps for Site 3.



6. SPONTANEOUS SUCCESSION OF RIPARIAN FYNBOS: IS UNASSISTED RECOVERY A VIABLE RESTORATION STRATEGY?

This chapter appeared as Reinecke *et al.* (2008) in a special issue of the South African Journal of Botany on Riparian Restoration, issue 74(3), and presents the results of the vegetation component of the Silvermine study. Because it is a published paper there is some repetition of wording and graphics, particularly in the introduction and methods sections. Appendix 2 presents a species list for plants collected at each site in each of the two years.

6.1 Abstract

The invasion of alien trees is a major threat to the freshwater resources and biodiversity of South Africa. The Working for Water (WFW) Program was initiated in 1995 in order to control the growth and spread of woody alien species in riparian zones, but the extent to which the indigenous vegetation naturally recovers following alien clearance remains poorly understood. In this study spontaneous succession of riparian vegetation following wild fires and alien clearing was monitored over a number of years at two sites on the Cape Peninsula: a pine plantation in an upland plateau and an *Acacia* spp.-invaded valley floodplain. After clearing, the vegetation at the pine site was successfully recovering along a trajectory towards Afromontane forest and as a result it is suggested that no active restoration is required. By contrast, our results show that areas cleared of *Acacia* spp. may be less resilient, with extensive regeneration of woody aliens and only a negligible recovery of indigenous trees. We propose that the absence of riparian trees may have been responsible for precipitating the transition to a community dominated by weedy nitrophilous grasses, and find evidence that this may be perpetuated through the continued removal of *Acacia* spp. It is hypothesized that this grass-dominated state may be resilient to natural restoration and thus represents an additional constraint to the recovery of riparian communities. Under such circumstances, we argue that active restoration would be required in order to re-instate the riparian community.

Key words: Alternative stable state; Biological invasions; Nitrophilous grasses; Restoration; Riparian zones; Spontaneous succession.

6.2 Introduction

The restoration of degraded ecosystems is recognised as an essential component in stemming the global loss of biodiversity (Hobbs and Harris 2001). As one of the primary drivers of species extinction (Vitousek *et al.* 1997) the control of invasive species is a central task of restoration ecologists (D'Antonio and Meyerson 2002). In common with riparian ecosystems throughout the world, the riparian vegetation of the Western Cape of South Africa has been heavily invaded by exotic species (Richardson *et al.* 1992). The most aggressive invaders in this region are trees and woody shrubs in the genus *Acacia.* These species out-compete indigenous plants and disrupt ecosystem functioning by altering the fire regime, patterns of sedimentation and erosion, the availability of water (Galatowitsch and Richardson 2005; Richardson *et al.* 2007) and soil nutrient levels (Yelenik *et al.* 2004).

Following the recognition that alien trees threatened the supply of freshwater resources, large areas previously invaded have been cleared under the auspices of the Working for Water (WFW) Program (van Wilgen *et al.* 1998). A number of clearance techniques are employed (see Blanchard and Holmes 2008), with follow-up treatments often implemented to remove regenerating aliens. In order to save costs, however, no active intervention to assist the recovery of indigenous species is carried out. The extent to which the indigenous vegetation recovers and resists re-invasion by woody aliens, is critical for the long term success of the WFW Program (Holmes *et al.* 2005).

Riparian community composition and structure are determined by, and in turn influence, a variety of factors including; channel morphology, discharge, flood regime (Naiman *et al.* 2005) and the frequency and intensity of fires (Kruger 1978; Sieben 2002). In the Western Cape, it has been hypothesized that site moisture levels and vulnerability to fires are key abiotic determinants of riparian community structure. In addition, soil type (texture and pH), creates a feed-back loop of community structure and composition on fire frequency by promoting certain growth forms of different flammability and resistance to fire. Western Cape riparian vegetation is dominated by broad-leaved woody species of scrub, predominantly perennial shrubs and small trees, but also includes characteristic fynbos elements such as species of Restionaceae and Ericaceae (Cowling and Holmes 1992). The understorey comprises forbs and graminoids, with continual recruitment of perennials and woody species occurring within the sub-canopy. In steep kloofs sheltered from fire, large tree species may establish leading to succession towards Afromontane forest (Taylor 1978). Few studies have investigated the resilience of riparian communities in the Western Cape to the invasion and clearance of alien trees (but see Blanchard and Holmes 2008; Vosse *et al.* 2008). There is evidence that the recovery of indigenous woody riparian species may be extremely limited, with extensive re-generation of woody aliens (Galatowitsch and Richardson 2005). This paper reports on the spontaneous succession of riparian vegetation following alien clearing in a single river catchment over a number of years. Although focussing on a single case study may trade generality for detail, analysis of vegetation change at a single locality through time may offer new insights into whether spontaneous succession of riparian communities in the Western Cape is a viable restoration strategy. For example, failure of woody riparian species to re-establish following clearance may be due to either a degenerate seed bank, a lack of propagule supply, or high levels of juvenile mortality associated with the clearing practices; each of which may require a different restoration strategy. The dynamics of spontaneous succession in the process of riparian vegetation recovery following alien removal thus warrants further investigation.

6.3 Study site

The Silvermine River catchment, located on the Cape Peninsula is contained within the boundaries of the Table Mountain National Park (Figure 6.1). Extensive stretches of the river were invaded or planted with a number of alien species, including *Pinus pinaster* D.Don, *Acacia saligna* (Labill.) H.L. Wendl., *Acacia longifolia* (Andrews) Willd., *Populus* X *canescens* (Ait.) Sm. (pro sp.), *Lantana camara* L. and *Paraserianthes lophantha* (Willd.). In January 2000, wild fires burned large areas of the Cape Peninsula, incinerating the majority of the vegetation in the Silvermine River catchment and leaving only a few large trees standing. The remaining alien trees and regenerating aliens were cleared by WFW personnel over successive follow up treatments. In line with WFW policies, no re-vegetation strategies were employed.

6.4 Methods

6.4.1 Data collection

Two sites were selected along the course of the Silvermine River. The first site had supported a *Pinus pinaster* (Cluster Pine) plantation in the mountain stream zone (hereafter Site 1), while the second was located in the foothill zone (hereafter Site 2) and had supported dense stands of *Acacia longifolia* (Long-leaved Wattle) and *A. saligna* (Port Jackson willow) (Reinecke and King 2003). Given that these two sites occur under different hydro-geomorphological settings (see Rowntree and Wadeson 1999), they represent alternative case studies and are not compared directly. Analysis of aerial photographs indicated that alien trees had formed closed canopies at these sites approximately 20 years prior to sampling.

At each site, two belt transects were used to survey the vegetation. These were positioned approximately 15 metres apart, and ran perpendicular to the river channel, extending into the mountain fynbos on each side of the valley. This allowed the riparian community to be delimited statistically on the basis of species composition rather than by eye. The four belt-transects consisted of contiguous metre-square plots within which the aerial cover of each species was estimated, and the maximum height of each species measured: between 49 and 78 plots were sampled along each transect. A permanent beacon cemented at each end of the transects enabled the same plots to be accurately resurveyed during summer (January through March) of year one, two and four following the fire in January 2000. All vegetation data were entered into the vegetation database TURBOVEG V1.99 (Hennekens 1996). Along each belt transect, the valley topography was surveyed using an electronic theodolite (Leica TC307 model) and a standard Leica prism and staff. The position of high water during the winter flood of July 2001 was recorded to aid in delineating the outer boundary of the riparian vegetation.

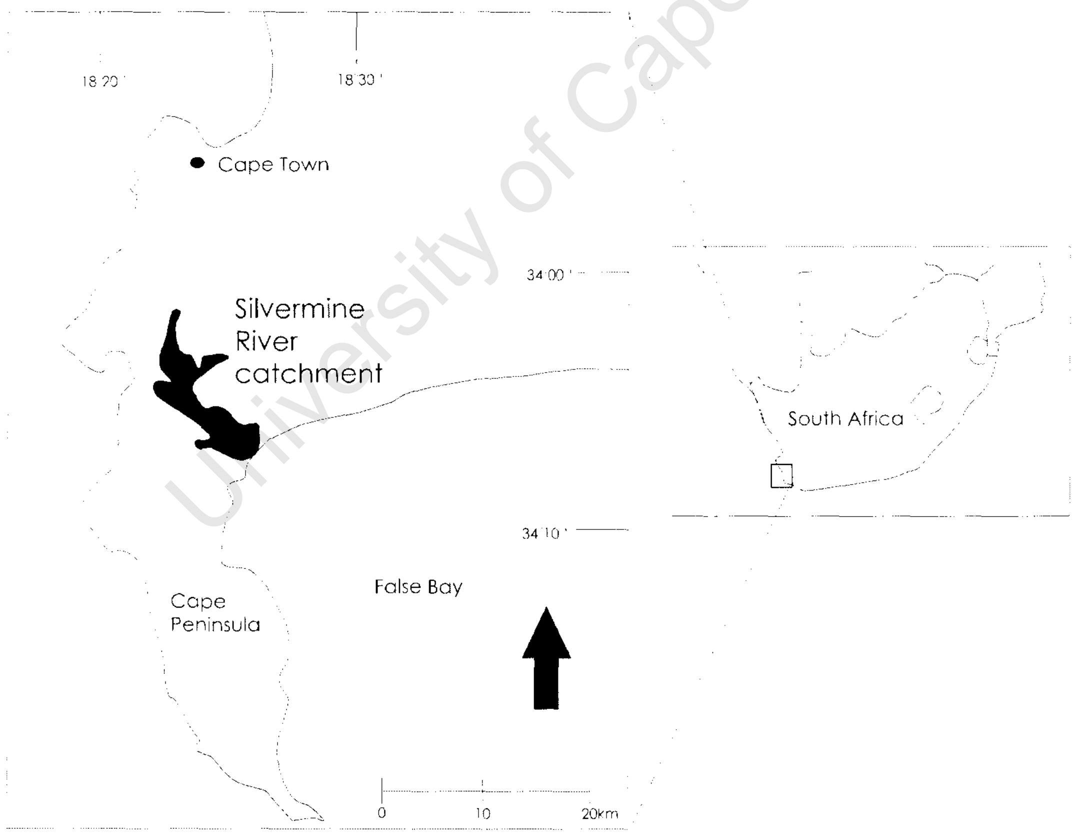


Figure 6.1 **Location of the Silvermine River catchment on the Cape Peninsula.**

6.4.2 Statistical analysis

Non-parametric statistics were used to delimit the riparian community. Species cover data were 4th root transformed to achieve a balance between the influences of common and rare species (Clarke and Warwick 2001). For each belt transect, the Bray-Curtis similarity coefficient was used to convert the cover data to a matrix of similarity between plots (a method which does not take into account joint absences). Similarity was then compared between plots using Non-metric Multi-Dimensional Scaling (MDS) graphs and CLUSTER dendrograms implemented in PRIMER (Clarke and Warwick 2001). Plots belonging to the riparian zone were identified as the contiguous plots that grouped together in both 2001 and 2002. These plots fell within the marked floodline on the surveyed cross-sections, validating delineation of the riparian community. At each site, the plots of the two belt transects that were identified as riparian were combined. The number of indigenous and alien species and their mean richness within each plot were calculated, and measures of species diversity and equitability were also calculated for indigenous species. In order to identify the dominant species at each site, species-cover data were ranked by frequency of occurrence (100% = a species occurred in all plots at a site) and total cover (the average over all plots within a site).

Given that funds for clearing are limited, the extra costs required for any active intervention, in order to fast track recovery, must be justified if the goal of alien clearing programmes is succession towards naturally functioning riparian communities that can buffer against further invasion. Plant species attributes and life history strategies are important in determining which riparian lateral zone a species may occupy and hence the structure and composition of different riparian communities (Holmes *et al.* 2005). Some preliminary work on community composition and structure was done by Reinecke *et al.* (2007) who hypothesized a trajectory of recovery for Riparian Scrub stands between one and ten years of age following alien clearing or fire. Their model described the frequency of growth forms at different life stages of common Riparian Scrub species on Western Cape riverbanks, some of which tended to occur at predictable elevations. They found that a description of the frequency of growth forms at different life stages was appropriate in order to characterise community structure and assess recovery. On this basis, each species was classified into one of the six growth forms (Table 6.1) according to (Goldblatt and Manning 2000), and the number of species and the total plant cover belonging to each group was summed. Vegetation structure also provides an important measure of community recovery, given its key role in regulating the susceptibility to fire and its hypothesized role in attracting vertebrate dispersers (Richardson *et al.* 2007). To investigate changes in vegetation structure, the maximum height of each species occurring within each plot was measured and assigned to one of four vertical strata (m): <0.5, 0.5-

2.0m, 2.0-5.0 and >5.0m (Galatowitsch and Richardson 2005). The total plant cover occurring in each height class was calculated for each site. The tallest stratum with more than 25% of the total vegetation cover was assigned as the dominant height class, and the number of occupied strata and relative cover in each was used to calculate a simple measure of Plant Height Diversity using the Shannon-Weiner diversity index. Although not all of the plants foliage may fall within the assigned stratum, for the purposes of this paper, this limitation was accepted.

Table 6.1 **The six growth forms used to classify species groups, with heights referring to the adult of the species.** Growth form nomenclature follows (Goldblatt and Manning 2000).

Growth form	Definition
1 forb	A broad leafed herbaceous plant other than graminoids.
2 graminoid	Plants in the family Juncaceae, Cyperaceae, Poaceae and Restionaceae.
3 ericoid	Shrubs with ericaceous leaves.
4 shrub	A low or medium sized woody perennial plant often with multiple stems (<1m).
5 small tree	A large woody perennial plant usually with multiple stems or with main trunk (2-10m).
6 tree	A tall woody plant with main trunk, branches and a distinct elevated crown (>10m).

Invasion of woody alien species into the Silvermine catchment was so extensive that few, if any, areas were suitable to serve as appropriate reference sites against which recovery could be assessed. Prior to the fires (January 2000), mature Afromontane Forest occurred along the river at Site 1, while at Site 2, the more open valley and susceptibility to fires suggested that riparian scrub would naturally predominate (Taylor 1978). Acknowledging the difficulties in assigning appropriate restoration targets, the recovery of the riparian vegetation was assessed by comparing the species composition of each site at four years post fire to a reference condition database for Riparian Scrub in mountain stream and foothill reaches of Western Cape rivers (Table 6.2, Reinecke *et al.* 2007).

At Site 2, there was a dramatic expansion in the cover of the perennial grass *E. setacea* between the 2001 and 2002 survey, coincident with the follow up clearance programmes that removed regenerating *Acacia* spp. A *post hoc* analysis of the change in percentage cover of *Acacia* spp. and *E. setacea* between 2001 and 2002 was performed, using ordinary least squares regression implemented in the R programming environment (R Development Core Team 2004). In order to meet with the assumptions of constancy of variance and normality of errors, the percentage data was arc-sine transformed prior to analysis (Crawley, 2002).

Table 6.2 **Species commonly found in Riparian Scrub and Afromontane Forest communities** (Reinecke *et al.* 2007). + indicates the species was present at one of the sites. ST = small tree, FO = forb, GR = graminoid, SH= shrub and TR = tree.

	Site			Site	
Riparian Scrub	1	2	Afromontane Forest	1	2
Brabejum stellatifolium (ST)			*Asparagus scandens* (FO)		
Brachylaena neriifolia (ST)			*Blechnum capense* (FO)		
Calopsis paniculata (GR)			*Blechnum punctulatum* (FO)		+
Cannamois virgata (GR)			*Canthium ventosum* (TR)		
Diospyros glabra (SH)			*Cunonia capensis* (TR)	+	
Elegia capensis (GR)			*Diospyros whyteana* (TR)		
Erica caffra (ST)			*Ehrharta erecta* (GR)		
Freylinia lanceolata (ST)			*Halleria lucida* (TR)		
Halleria elliptica (SH)			*Histiopteris incisa* (FO)	+	+
Juncus lomatophyllus (GR)			*Ilex mitis* (TR)		
Metrosideros angustifolia (ST)			*Juncus effusus* (GR)	+	+
Morella serrata (ST)			*Kiggelaria africana* (TR)		
Prionium serratum (SH)		+	*Podocarpus* sp. (TR)		
Psoralea pinnata (ST)		+	*Pteridium aquilinum* (FO)	+	+
Pteridium aquilinum (FO)	+	+	*Rapanea melanphloeos* (TR)		
Rhus angustifolia (ST)			*Secamone alpinii* (FO)		
Salix mucronata (ST)			*Todea barbara* (FO)	+	
Todea barbara (FO)	+		*Virgilia oroboides* (TR)	+	

6.5 Results

Using CLUSTER analysis and MDS ordination, 18 plots at Site 1 and 32 plots at Site 2 were demarcated as belonging to the riparian communities (Figure 6.2). Vegetation recovery was highly disparate between the two sites, with differences in the relative contributions of the various growth forms, the identity of the dominant species, and the resulting vegetation structure.

6.5.1 Cover and species richness of indigenous and non-indigenous species

Site 1

In the immediate post-fire year, 13 indigenous species were recorded in the riparian zone, with a mean richness per plot of 1.94±1.55/m^2 (Table 6.3). Between 2001 and 2004, the total number of indigenous species remained approximately constant but there was a consistent increase in mean plant cover (Figure 6.3a) and species richness per plot. Levels of diversity declined, reflecting the increased dominance of a few common species. The almost constant total indigenous species richness over the 4-year study period masked a high level of turnover in species composition, with only 50%-60% of species shared between successive years (Table 6.4). Seven alien species were present in the immediate post-fire year at low cover. The cover of aliens increased up to the 2002 survey but subsequently declined to low levels by 2004.

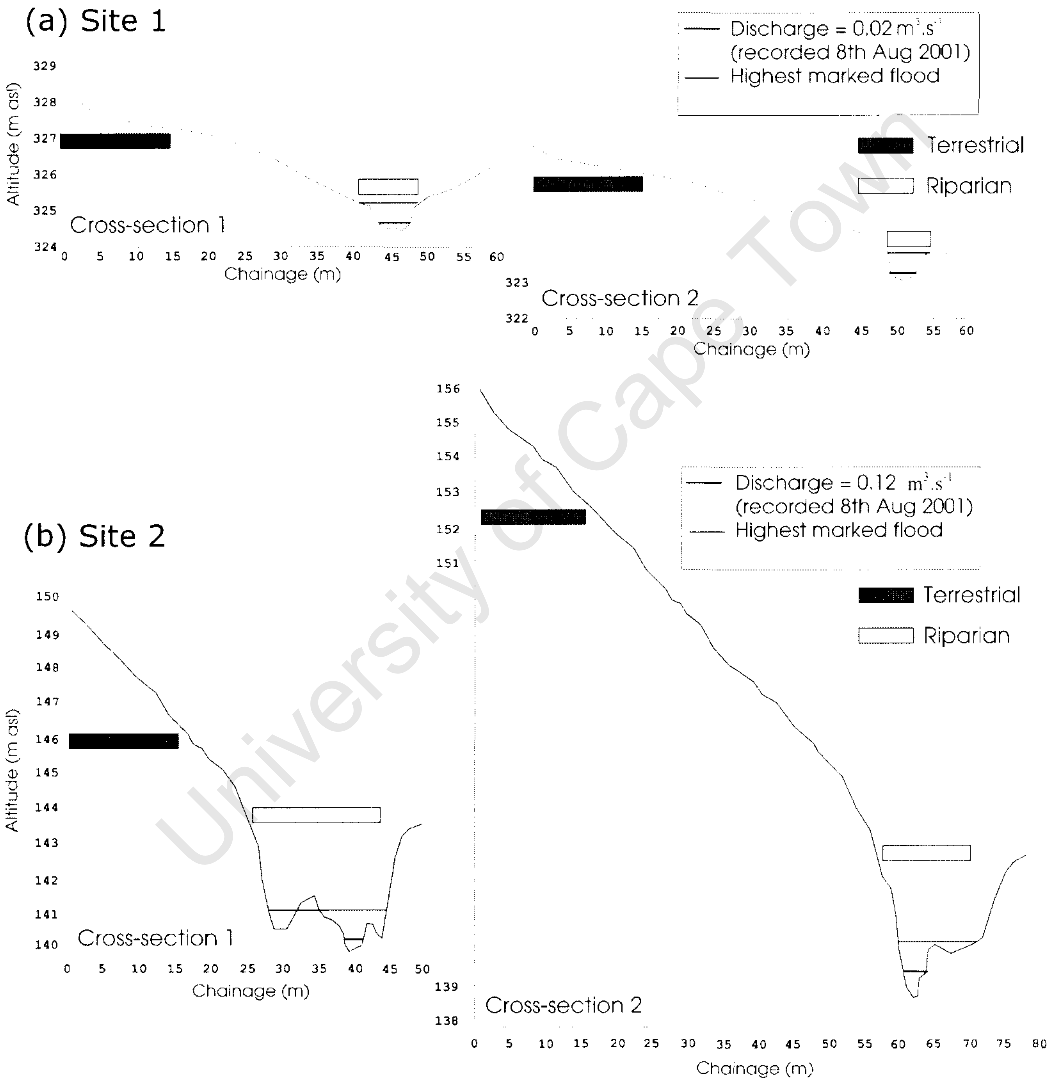


Figure 6.2 **Cross-sections (surveyed in summer 2001) depicting the channel shape surveyed at Sites 1 (a) and 2 (b).** Vegetation transects were sampled three times over four years (2001/2/4) along the length of each cross-section at each site. The location of summer low flow, the position of the winter high flow flood line and the location of terrestrial and riparian plots are indicated. Riparian plots are located within the flood line in both cases.

Table 6.3 **Descriptive biodiversity and vegetation statistics at both sites for each year.** At site 1, the cover of indigenous vegetation and richness increased from 2001 to 2004. Species richness, diversity and equitability remained relatively constant and there was no increase in plant height diversity and dominant height class over the four years at site 1. At site 2, the removal of the regenerating *Acacia* spp. between 2001 and 2002 changed the structural complexity from 0.95 to 0.71. Due to a lack of indigenous woody species the vegetation structure remained stunted (dominant height class 0.5-2m) and there was lower plant height diversity over the four years. ± indicates SD of the mean.

	Site 1 (plots=18)			Site 2 (plots=32)		
Year	2001	2002	2004	2001	2002	2004
Species Richness (n)						
Total	20	21	18	29	31	28
Native	13	16	13	26	24	24
Alien	7	5	5	3	7	4
Mean Species Richness (n)						
Total	3.00±2.11	3.50±2.50	3.44±1.92	5.03±3.4	3.91±2.57	3.69±2.31
Native	1.94±1.55	2.56±1.76	2.72±1.45	3.78±3.15	3.03±2.07	3.28±2.16
Alien	1.06±0.87	0.94±1.26	0.72±1.01	1.25±0.72	0.88±0.98	0.41±0.76
Mean vegetation cover (cm^2)						
Total	26.70±34.3	63.40±53.7	83.70±44.3	62.30±57.2	65.60±39.3	92.60±54.3
Native	22.60±29.4	43.00±36.3	71.70±37.6	44.40±45.2	58.60±38.1	90.30±55.9
Alien	4.10±5.8	20.40±35.4	12.00±25.2	17.80±24.6	6.90±12.4	2.30±6.1
Shannon-Weiner						
Species Diversity (H)	2.49	2.59	2.28	2.29	2.48	2.37
Equitability (E)	0.83	0.85	0.79	0.680	0.72	0.71
Simpson's Index						
Species Diversity (H)	9.52	11.49	7.67	7.02	5.78	6.35
Equitability (E)	0.48	0.55	0.43	0.24	0.19	0.23
Plant height diversity (H)	0.97	0.96	1.35	0.95	0.71	0.84
Dominant height class (m)	0.5-2	0.5-2	>5	0.5-2	0.5-2	0.5-2

Table 6.4 **Sørensons similarity index reflecting turnover in species composition over four years.** There was a high level of turnover in species composition, with only 50%-60% of species shared between successive years, although species richness remained fairly constant (Table 6.3).

Years compared	Site 1	Site 2
2001×2002	0.63	0.67
2002×2004	0.51	0.61
2004×2002	0.53	0.56

Site 2

There was a rapid increase in plant cover in the first year following the fire (Figure 6.3b). Although predominantly driven by indigenous species (Table 6.2), alien species also contributed substantially. In the following year, total plant cover increased only marginally due to a decline in the cover of alien species balancing the increase in the cover of indigenous taxa. The cover of aliens continued to decline to 2004, while there was large increase in the cover of indigenous species. There was little change in species richness (measured at either the site or plot level), diversity and equitability over the study period.

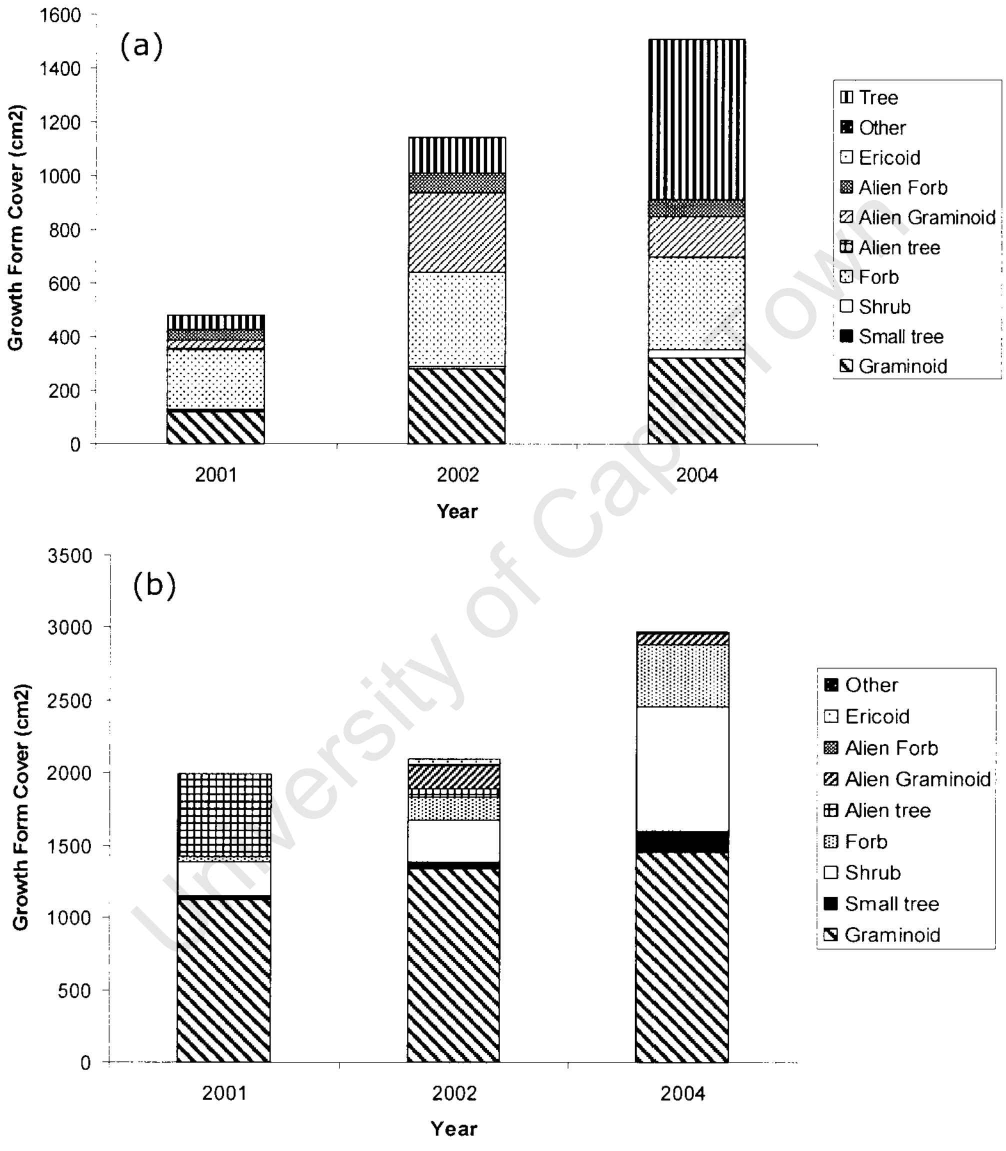


Figure 6.3 **Changes in the total growth form cover between 2001 and 2004 for all riparian plots at (a) Site 1 and (b) Site2.** At Site 1, non-indigenous annual graminoids peaked and subsequently declined, while there were steady increases of indigenous trees, forbs and graminoids. In contrast, at Site 2, there was an immediate increase in perennial graminoids and alien trees in the immediate post fire environment. There was a gradual increase in the cover of shrubs and forbs, while perennial graminoids remained dominant.

6.5.2 Trends in growth form and species composition

Site 1

In the first year following fire, the recovering vegetation was dominated by indigenous forbs and graminoids (Figure 6.3a). A number of ephemeral alien species including *Taraxacum officinale* and *Briza maxima* were present throughout the site. Numerous indigenous tree seedlings successfully germinated from the seed bank including; *Cunonia capensis, Virgilia oroboides, Curtisia dentata* and *Psoralea pinnata*. By contrast, only a few alien *Pinus pinaster* seedlings were present and these were subsequently removed by follow up clearing treatments.

In the following year, there was a large increase in the cover of large trees due primarily to the rapid growth of *Virgilia oroboides* seedlings. Two indigenous tree species, a large and small tree respectively, *Curtisia dentata* and *Psoralea pinnata,* were present in 2001 but absent by the 2002 survey. By 2004, the community was characterised by a dense canopy of large *Virgilia oroboides* and *Cunonia capensis* trees, with an understorey of graminoids and re-sprouting forbs. Six of the most common species present are considered to be typically Afromontane (Table 6.2), while there was an obvious lack of ericoids, restioids, shrubs and small trees (Figure 6.3a). There was a high level of plant structural complexity, and a dominant height class of >5m (Table 6.3).

Site 2

The rapid regeneration in the immediate post-fire year was driven by perennial graminoids and alien *Acacia* spp. which accounted for more than 50% and 25% of the vegetation cover respectively (Figure 6.3b). *Acacia* spp. were ubiquitous throughout the site (*A. longifolia* alone was present in over 80% of plots) and formed a closed canopy that rapidly overshadowed the other species; some individuals growing to 2.5 metres in height in a single year. In contrast, there was only a limited regeneration of a few small indigenous tree species scattered across the site (*Erica caffra* and *Psoralea pinnata*) and no large tree species.

Follow-up clearances between the 2001 and 2002 surveys removed most of the regenerating *Acacia* spp. Coincident with this was a large increase in the cover of *Ehrharta setacea*, which by 2002 comprised approximately 40% of the total plant cover (total graminoid cover increased only marginally because of the death or decline of a number of other species). Plots with the greatest reduction in cover of *Acacia* spp. during this period exhibited significantly greater increases in cover of *E. setacea* ($r^2 = 0.3$, slope = -0.64, $p = 0.001$, Figure 6.4).

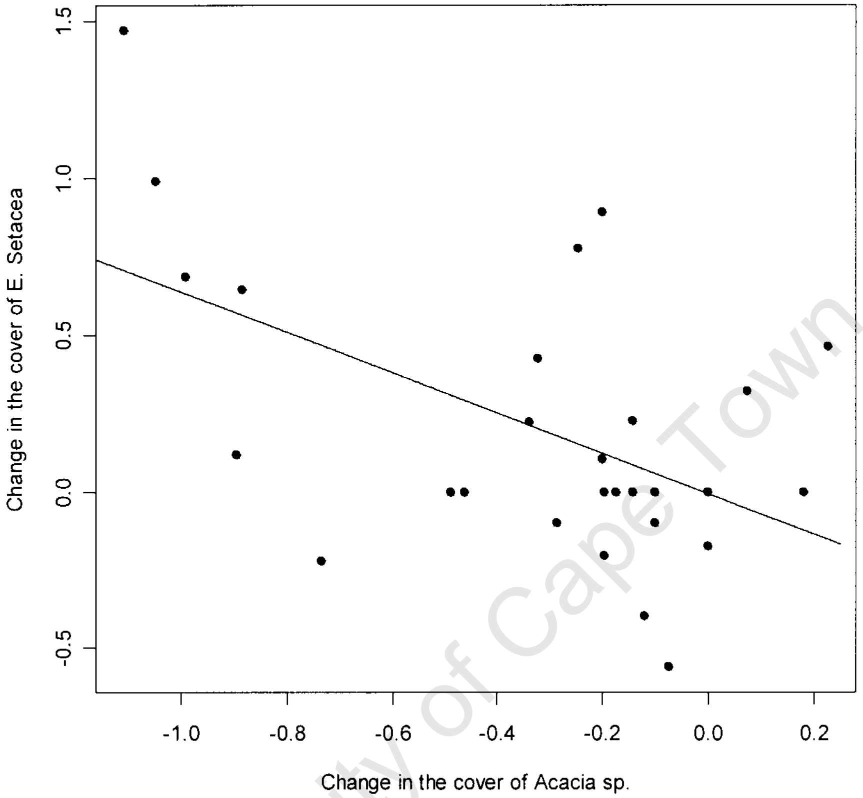


Figure 6.4 **Comparison of *Acacia saligna* and *A. longifolia* cover (arc-sine transformed) and corresponding growth of the perennial grass *Ehrharta setacea* between 2001 and 2002.** As cover of *Acacia* spp. was reduced, the cover of perennial grasses increased significantly, ($r^2 = 0.3$, slope = -0.64, $p = 0.001$).

During the study period, there was a gradual increase in the cover of shrubs and forbs, of which the most common species were; *Berzelia lanuginosa*, *Oftia africana*, *Pteridium aquilinum* and *Vellereophyton dealbatun*. The cover of indigenous small trees increased only marginally, while by 2004 large indigenous tree species remained completely absent from the site. Furthermore, the small tree guild was represented by only a single species, *Psoralea pinnata* (Figure 6.3a): this species has been categorized by previous authors as either a small tree, or as a shrub by Goldblatt and Manning (2000). The only other member of this guild, *Erica caffra,* had perished. The increase in cover of the shrub and small indigenous tree guild was driven by the growth of those individuals that had initially regenerated, rather than continued recruitment.

Reflecting the lack of indigenous woody species, there was no increase in the dominant height class over the study period, remaining at 0.5-2m. Plant height diversity declined between 2001 and 2004 due to the removal of regenerating *Acacia* spp. (Table 6.3). Reinecke *et al.* (2007) re-sampled the site in 2005 and found that, with the exception of *Psoralea pinnata*, indigenous trees remained absent from the site. Perennial grasses on the other hand, such as *Ehrharta setacea*, maintained a high cover and frequency.

6.6 Discussion

6.6.1 Temporal patterns in post-clearance regeneration

Analysing the temporal patterns of community assembly, following the clearance of alien invasive trees, provided a number of insights into the dynamics of riparian communities with implications for their management and restoration. It was apparent that all species of indigenous trees (large and small) and shrubs, the dominant guilds in riparian scrub, which emerged from the seed bank did so within the first few years following clearing (also see Vosse *et al.* 2008). This suggests that despite the relatively long time frame of community development expected in such systems (Reinecke *et al.* 2007), it is possible to assess the likely trajectory of recovery and determine the need for active restoration shortly after the initial clearing operations. Indeed, within our four-year study period, it was evident that the recovery of vegetation at the two study sites was proceeding along markedly different trajectories.

Where *Pinus pinaster* was cleared, the riparian vegetation recovered rapidly forming a community dominated by large indigenous trees with an understorey of graminoids and re-sprouting forbs. Although the community was relatively depauperate, lacking a number of species characteristic of Afromontane forest (Table 6.4), the high structural complexity of the vegetation, dense canopy cover, and stable substrata should provide optimal conditions for the immigration and establishment of these taxa (Galatowitsch and Richardson 2005). As a result, we suggest that no active restoration will be required at this site.

The burning of the dense stands of *A. saligna* and *A. longifolia* was followed by germination of woody aliens *en masse* from the soil-stored seed banks. Few indigenous tree seedlings were present in the post-fire community, which was instead dominated by herbaceous species and, to a lesser extent, shrubs. This supports the previous findings of Galatowitsch and Richardson (2005) and suggests that the limited resilience of riparian trees to prolonged invasion and clearance operations is largely due to their absence from the soil-stored seed bank. Our results also show however that a high mortality of juvenile riparian small trees may contribute to their rarity following alien clearance (see below).

Although the accelerated increase in the cover of the small tree and shrub guild between 2002 and 2004 suggests that woody species are becoming a more prominent component of the community at Site 2, this was driven by the growth of the few individuals that had initially regenerated. Consequently, as their growth declines, we expect future increases in the cover of these guilds to be minimal. Given that the seeds of many riparian trees are dispersed by birds, the low structural complexity of the vegetation resulting from a lack of woody species may inhibit the potential for future recovery and render this site susceptible to chronic re-invasion of woody aliens.

Development of the vegetation at both sites was characterised by a high level of turnover in species composition. Although this was principally driven by ephemeral species, our study also identified the enigmatic loss of a number of potentially long-lived woody taxa (*Curtisia dentata*, *Psoralea pinnata* and *Erica caffra*). The most likely explanation for the coincident mortality of these species is that they were removed during the follow-up clearance operations. Given the rarity of woody riparian taxa in the post-clearance community, the removal of only a few individuals may have a disproportionate impact on future vegetation recovery. Ensuring that WFW teams are adequately trained in the removal of alien species must be a priority as this will reduce the need for active re-vegetation, for which the costs may be prohibitive.

6.6.2 Secondary grass invasion and the prospects for future recovery

In addition to the absence of characteristic woody riparian species, the most notable feature of the vegetation at the site cleared of *Acacia* spp. was the exceptionally high cover of perennial graminoids, particularly *E. setacea*. Although this species is native to the fynbos biome, it does not comprise a dominant component in undisturbed riparian communities (Reinecke *et al.* 2007, Table 6.4). The trend for weedy perennial grasses to proliferate following the removal of *Acacia* spp. has been documented in terrestrial fynbos communities and has been attributed to the nutrient enriched soils resulting from nitrogen fixation by the *Acacia* spp. (Yelenik *et al.* 2004). A similar trend of nutrient poor shrub lands shifting to a grass dominated state has been widely documented for European heathlands subjected to atmospheric nitrogen deposition (Bobbink *et al.* 1998).

In contrast to nutrient poor terrestrial fynbos, riparian zones are noted for their relatively fertile soils. Indeed this is thought to accelerate the growth of indigenous trees and the development towards forest (Manders *et al.* 1992). The higher availability of nutrients would also be expected to promote the growth of perennial grasses, and yet under natural conditions this guild typically forms only a minor component of the initial post-fire riparian community (Reinecke *et al.* 2007). Given this context, elevated levels of soil nitrogen resulting from *Acacia* spp. invasion would seem to provide an inadequate explanation for the abundance of weedy grasses documented in this study. Instead we suggest that following fires the regeneration of Riparian Scrub and the associated reduction in light availability act to suppress the growth of perennial grasses. Where disturbance has been intense however, and the recovery of Riparian Scrub is limited, higher light availability can lead to the rapid increase in the cover of grasses. The significant association between the expansion in the cover of *E. setacea* and the removal of regenerating *Acacia* spp. supports this.

Despite mean *Acacia* spp cover in 2001 being 'only' 21%, the reduction in *Acacia* spp. cover explained approximately 30% of the variance in the change in *E. setacea* cover. We suspect that had regeneration of the *Acacia* spp. been more advanced, the effects of their removal would have been much more pronounced. This raises the possibility that follow-up clearing programmes may inadvertently perpetuate the dominance of weedy grasses, adding to a growing body of literature regarding the unintended, and often undesirable, consequences of attempting to control invasive species (Zaveleta *et al.* 2004).

The proliferation of perennial grasses in systems where they have previously been rare can lead to dramatic changes in community dynamics (Levine *et al.* 2003). For example, exotic grasses may out-compete indigenous tree seedlings (D'Antonio and Mack 2001) and, through their rapid growth rates and availability of fine fuel, may initiate a self-perpetuating cycle of invasion by promoting more frequent fires (D'Antonio and Vitousek 1992). It is unlikely that the high cover of perennial grasses was a factor limiting the recovery of other riparian elements in this study, since the regeneration of most species took place prior to the expansion of *E. setacea*. However, given the dominance now attained by perennial grasses, we suspect that their regeneration from meristems and the soil seed bank, following future fires, is likely to be both more pronounced and rapid (Milberg and Lamont 1995). This may adversely effect the seedling recruitment of woody riparian species decreasing the diversity of the community over successive fire cycles. Furthermore, given that Riparian Scrub development is inhibited by short fire cycles (Galatowitsch and Richardson 2005) the possible effects of *E. setacea* on fire dynamics may be particularly important.

Through such self-reinforcing feedbacks (Suding *et al.* 2004), invasion by perennial grasses into recently-cleared areas may represent an additional constraint to those identified by Galatowitsch and Richardson (2005) already inhibiting Riparian Scrub recovery. If the key factor precipitating the dominance of perennial grasses is the absence of riparian trees, we recommend re-instating this guild in order to catalyze recovery. In contrast to the measures proposed for the fynbos (Yelenik *et al.* 2004), reducing nutrient levels in riparian ecosystems naturally characterised by fertile soils would seem neither desirable nor achievable and is unlikely to succeed in reducing the dominance of perennial grasses. Additionally, the potential for perennial grasses to inhibit woody seedling survival suggests that the planting of indigenous trees may be more successful at catalysing recovery than sowing seeds in areas where the cover of perennial grasses is high. Future research should attempt to identify both the factors promoting the proliferation of weedy grasses and their effects on the regeneration of both indigenous and alien woody species.

6.7 Acknowledgments

The authors gratefully acknowledge many students involved in data collection and processing. In particular, Dr David Miller is thanked for many hours of field work, and statistical advice. Certain specialists provided input during the study and Dr Neil Armitage, Professor Malcolm Newson, Professor Kate Rowntree and Dr Charlie Boucher are thanked for their input. Particular thanks to Dr Patricia Holmes for her thorough botanical critique during analysis of these data. The Water Research Commission (WRC) funded the research and Dr Steve Mitchell is thanked for his role are project manager. The Zoology Department at the University of Cape Town housed the project.

7 SPONTANEOUS RECOVERY OF AQUATIC MACROINVERTEBRATES

7.1 Introduction

Riverine invertebrates are critical components of freshwater ecosystems, contributing to aquatic and terrestrial food chains, the processing of lotic organic matter and the purification of water. Lotic systems are naturally heterogeneous, with variability in habitat occurring at multiple spatial and temporal scales (Dallas and Day 2007). Aquatic macroinvertebrates are assumed to distribute themselves amongst the available aquatic habitats (biotopes, see Section 2.4; Rowntree 2001) according to optimal space (quality and quantity) and time (seasonal) requirements (Snaddon 2007). Invertebrate assemblages thus adjust continually in response to persistent and fluctuating environmental factors. As a result, macroinvertebrate communities are often patchily distributed (Pringle *et al.* 1998).

Western Cape rivers within the Cape Floristic Kingdom are winter rainfall systems that are acidic and typically poor in dissolved nutrients. Summer low flows and droughts contrast with stochastic and intense winter floods. Endemism is as high as 64% (Harrison and Agnew 1962; Picker and Samways 1996; Wishart and Day 2002) and has been attributed to these unique conditions, when compared to rivers in the rest of South Africa. Since upland reaches are generally less disturbed than lowland reaches they boost overall catchment biodiversity (Furse 2000, cited by Dallas and Day 2007). In the Western Cape, aquatic macroinvertebrate communities display what King and Schael (2001) have called 'catchment signatures'. They demonstrated that invertebrate samples from a broad spectrum of Western Cape rivers grouped together based upon similarities in community composition at varying levels (Schael and King 2005; Schael 2006). The first level of biotic groupings occurred at the catchment level and was speculated to be due to subtle physico-chemical signals or historical biogeographical patterns. The second level of biotic groupings was by channel type, with invertebrates from bedrock channels grouping separately to those from alluvial channels. The third determinant of biotic groupings was the location of the collected invertebrates along the river's longitudinal profile (see Section 2.2).

Fowler and Harding (2000) collected macroinvertebrates along the Silvermine River six months after the mountain fires of January 2000 (section 3.1) and assessed the river's Ecological Importance and Sensitivity based on the taxa present. Ecological Importance is an expression of the degree to which ecological diversity and functioning on a wider scale are maintained, whereas Ecological Sensitivity reflects the ability of an ecosystem to resist and recover from disturbance. The mountain stream (Site 1) had a 'very high' Ecological Importance and Sensitivity, whereas the foothills (Sites 2 and 3) had a 'high' rating. The 'very high' rating in the mountain stream was attributed to the presence of rare and unique taxa that are sensitive to changes in flow, habitat type and water quality, such as the Amphipoda (*Paramelita* spp.), stoneflies (Plecoptera,) and helodid beetle larvae (Fowler and Harding 2000). The foothill region was considered to be important as a refuge and corridor for the cape clawless otter (*Aonyx capensis*).

In this study data were collected over two successive years. Since the entire catchment was disturbed after the fires, no part of the catchment could act as a reference of baseline or undisturbed conditions for elsewhere in the catchment. A range of local methods that make use of macroinvertebrates for water-quality monitoring were used to assess changes in the communities of aquatic macroinvertebrates over the two year period (see Section 4.3).

7.2 Results

7.2.1 Investigating community composition

The overall relationship between all the invertebrate samples collected for this thesis (i.e. from three sites over two years) was investigated initially using a CLUSTER analyses and a MDS ordination (see section 4.3). The output from this analysis is a dendrogram (Figure 7.1) that portrays the Bray-Curtis similarity between samples. In a CLUSTER analysis, samples with a similar taxonomic affinity for one another, in this case based upon community composition to family level, group together at certain differing percentages of similarity. Different sample groups thus represent samples with similar community compositions.

In the first division, every sample from the mountain stream (Site 1; see Section 3.2) separated as group 1 (Figure 7.1) from the two foothill Sites (2 and 3). The rest of the samples, forming groups 2 to 6 and from Sites 2 and 3, were mixed together and less obviously separated by site. The coarse habitat units and the year in which the sample was taken appeared to influence the formation of groups 3, 4 and 5, while group 6 appeared to contain mixed samples with no obvious affiliation for one another.

Group 2 consisted of samples from fast-flowing areas on larger substrata such as cobble, boulder or bedrock. Samples in group 2 were not separated further by year or site. Whilst other samples from fast-flowing areas were represented elsewhere on the dendrogam, groups 3 to 6 consisted mostly of samples from slow-flowing areas with mixed substrata.

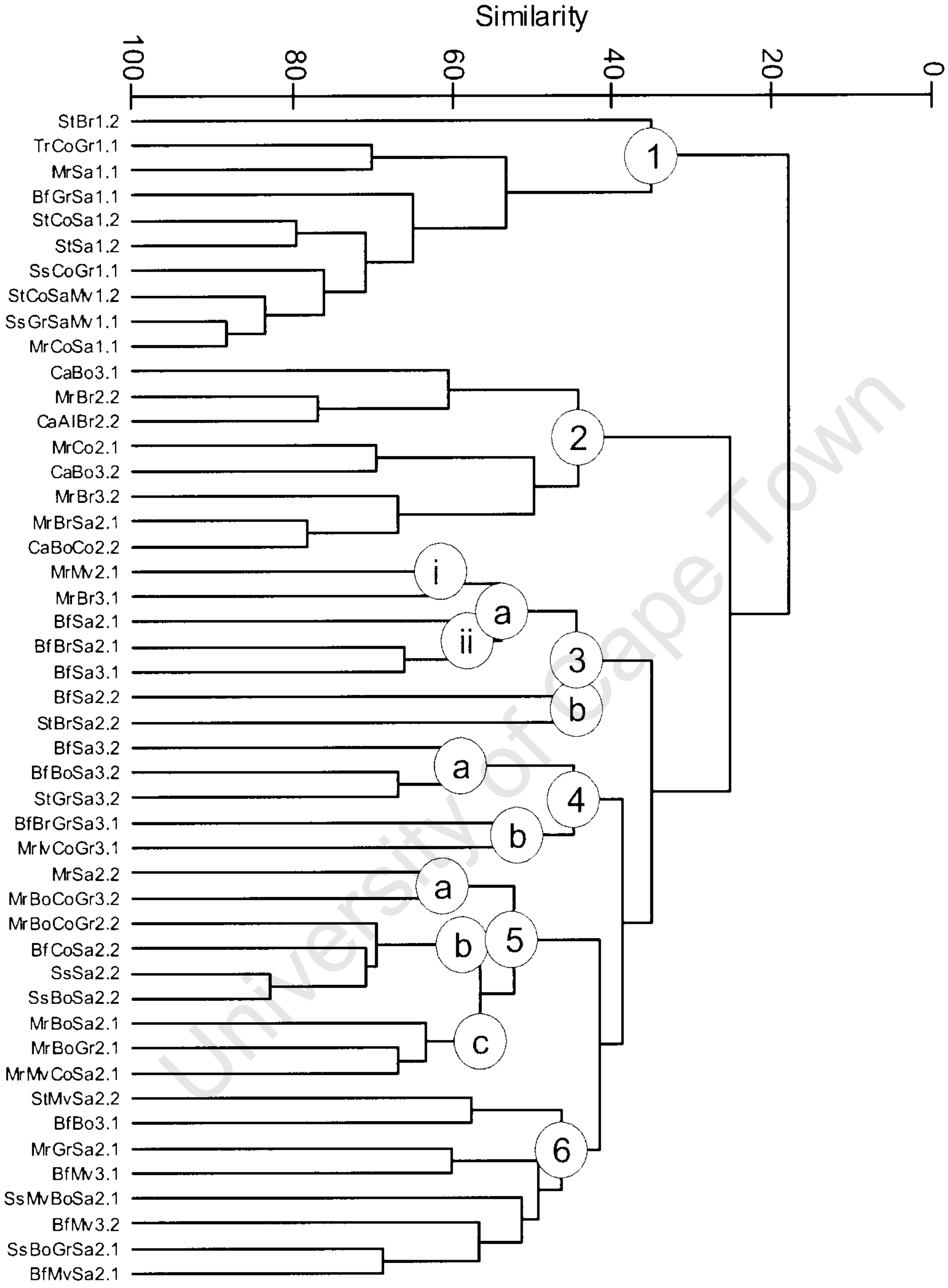


Figure 7.1 **CLUSTER analysis of Bray-Curtis similarity between all aquatic macroinvertebrate samples from Sites 1, 2 and 3.** Sample groups are formed from samples with a high percentage similarity, based upon the community composition. Sample codes are biotope components (as per Table 4.2), followed by site # and then sample year. For example the first sample of group 1, (StBr1.2) is a sample from a Still/Bedrock biotope at Site 1 collected in year 2.

In group 3 there were two main sub-groups. Group 3a consisted of samples from both Sites 2 and 3 that were collected in year 1, while group 3b consisted of two year 2 samples only. Group 3a split further into two sub-groups. Group 3ai with samples from fast-flowing areas and group 3aii with samples from slow-flowing sandy areas.

Group 4 consisted exclusively of samples from Site 3 that split into two sub-groups. Group 4a consisted of samples from slow-flowing areas with a sandy substratum from year 2 while Group 4b contained samples with two different sorts of biotopes from year 1.

Similarly group 5 split into three sub-groups. Group 5a with samples from fast-flowing areas on mixed substrata from Sites 2 and 3 both collected in year 2, Group 5b with three samples from slow-flowing areas and one from fast-flowing areas from Site 2 and year 2 only, and then group 5c with samples exclusively from fast-flowing areas from Site 2 all collected during year 1.

It appears that to some extent groups of samples have formed first on the basis of site (location), as sites from two adjacent zones separated from one another and then, for Sites 2 and 3 (groups 2 to 6 in Figure 7.1), on the basis of coarse habitat units, such as flow or substratum type, or the year in which the sample was collected. The CLUSTER and MDS analyses were repeated for the combined data sets from the two years for each site separately. If the results of a sister analysis, the non-metric multi-dimensional ordination (MDS) concurred with those of the CLUSTER analysis, the portrayal of the sample relationships was considered to be confirmed (Clarke and Warwick 2001).

Site by site analysis

Site 1

At Site 1, there was good separation of samples based upon their substratum and flow characteristics as shown by both the MDS and CLUSTER analyses (Figure 7.2). The one exception being sample # MrCoSa.1, a sample from a fast-flow area that separated out with those found in slow-flowing areas. Group 1 consisted of two samples from fast-flowing areas collected in year 1, with coarse and fine substrata respectively. Group 2 consisted of a mixture of year 1 and year 2 samples split into two sub-groups. Group 2a contained two samples from slow-flowing areas in year 2, one on a mixed substratum and the other on sand. Group 2b was a mixed group with three kinds of samples. There was a lone sample from a slow-flowing area on a coarse substratum, two from slow-flowing areas on mixed substrata and with marginal vegetation, and finally a single sample from a fast-flowing area on a mixed substratum.

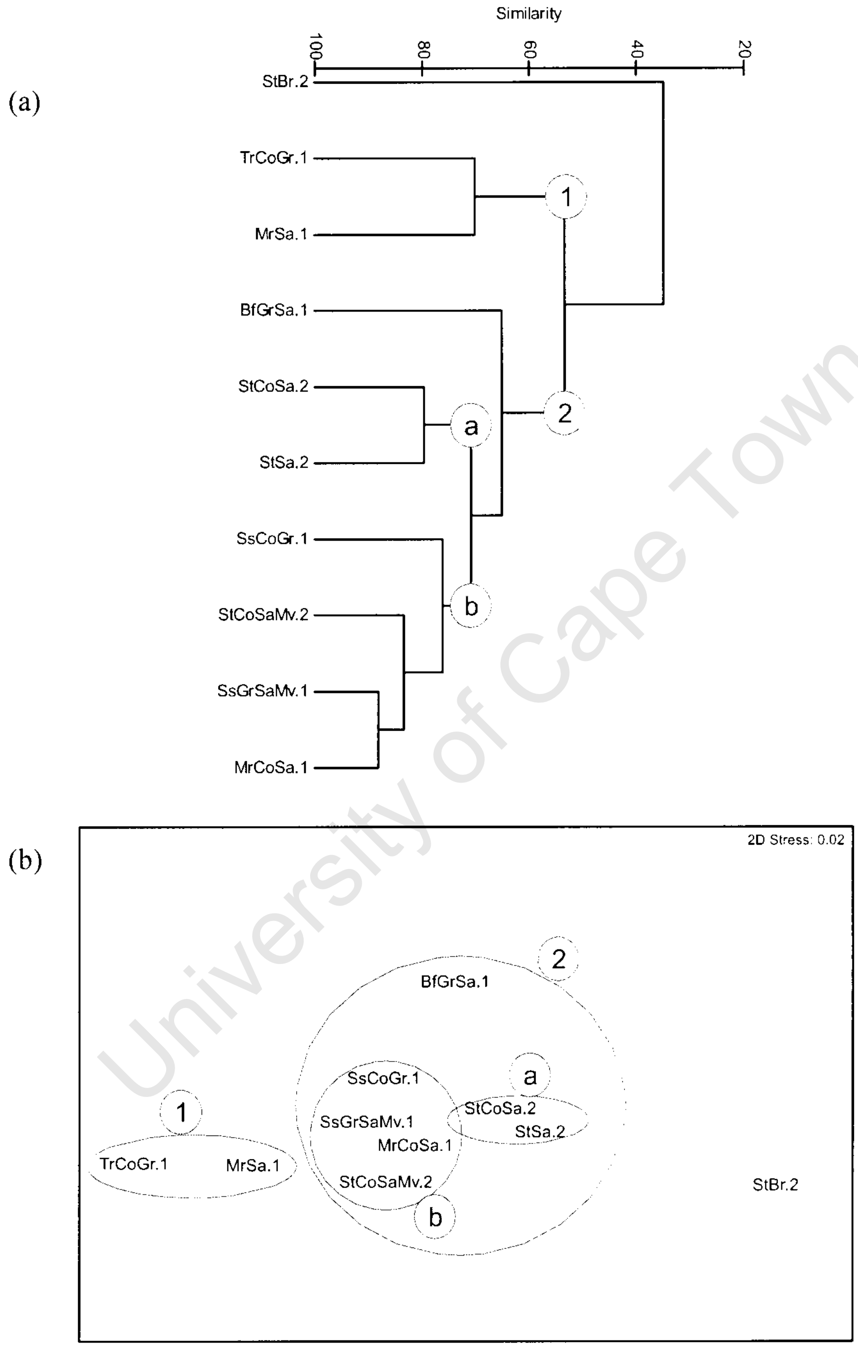


Figure 7.2 **(a) CLUSTER and (b) MDS analysis of Bray-Curtis similarity between aquatic macroinvertebrate samples from Site 1.** Sample groups are formed from samples with a high percentage similarity, based upon the community composition. Sample codes are biotope components (as per Table 4.2), followed by sample year.

Although there was to some extent separation by year or coarse habitat unit, the pattern is not clear, perhaps due to the few samples collected at this site. The same process was followed for the aquatic macroinvertebrate samples from the foothill Sites (2 and 3) lower down in the valley floodplain. More samples were collected at the two downstream sites so clearer patterns of separation, based upon the coarse habitat units and the year of the samples, were expected.

Site 2

Four groups were formed from the samples collected at Site 2 (Figure 7.3), with group 1 that contained two samples from fast-flowing areas on bedrock from year 2, separating from the other groups (2 to 4). Group 2 consisted of four samples from fast flowing areas split into two sub-groups with each sub-group containing a sample from each year. The two samples in group 2a both contained mixed coarse and fine substrata, while group 2b contained two samples from un-mixed substrata; one fine and one coarse. The pattern is less clear for group 3 with groups 3a and 3b containing samples of mixed flow and substratum types from both years: the single common factor for Groups 3a and 3b being the presence of sand, either alone or mixed with other substrata. Group 3c is more coherent on the CLUSTER than the MDS diagram so the validity of this sample group may be spurious. Nonetheless, group 3c contains three samples from fast flowing areas on a rocky and sandy substratum from year 1. Group 4 consists of three sub-groups of mixed habitat units. Group 4a contains mixed samples collected during year 1; group 4b contains two marginal vegetation samples, one from both years; while group 4c contains two samples from slow-flowing areas from year 2 with different substrata.

The samples of Site 2 were also not particularly well separated according to coarse habitat units or the sample year. This is most obvious in the MDS diagram (Figure 7.3) where the boundaries of sample groups 2, 3 and 4 overlap.

Site 3

In contrast to the analyses of Sites 1 and 2, the samples from Site 3 separated into three groups based upon the year of collection (Figure 7.4). Group 1a contained three samples from slow-flowing areas from year 2, each of which had sand as a substratum component. Group 1b contained a sample from each of slow and fast-flowing areas, from year 1 and 2 respectively, with no pattern evident in the substratum type. Group 2a contained two samples from slow-flowing areas; one from the marginal vegetation and the other over sand.

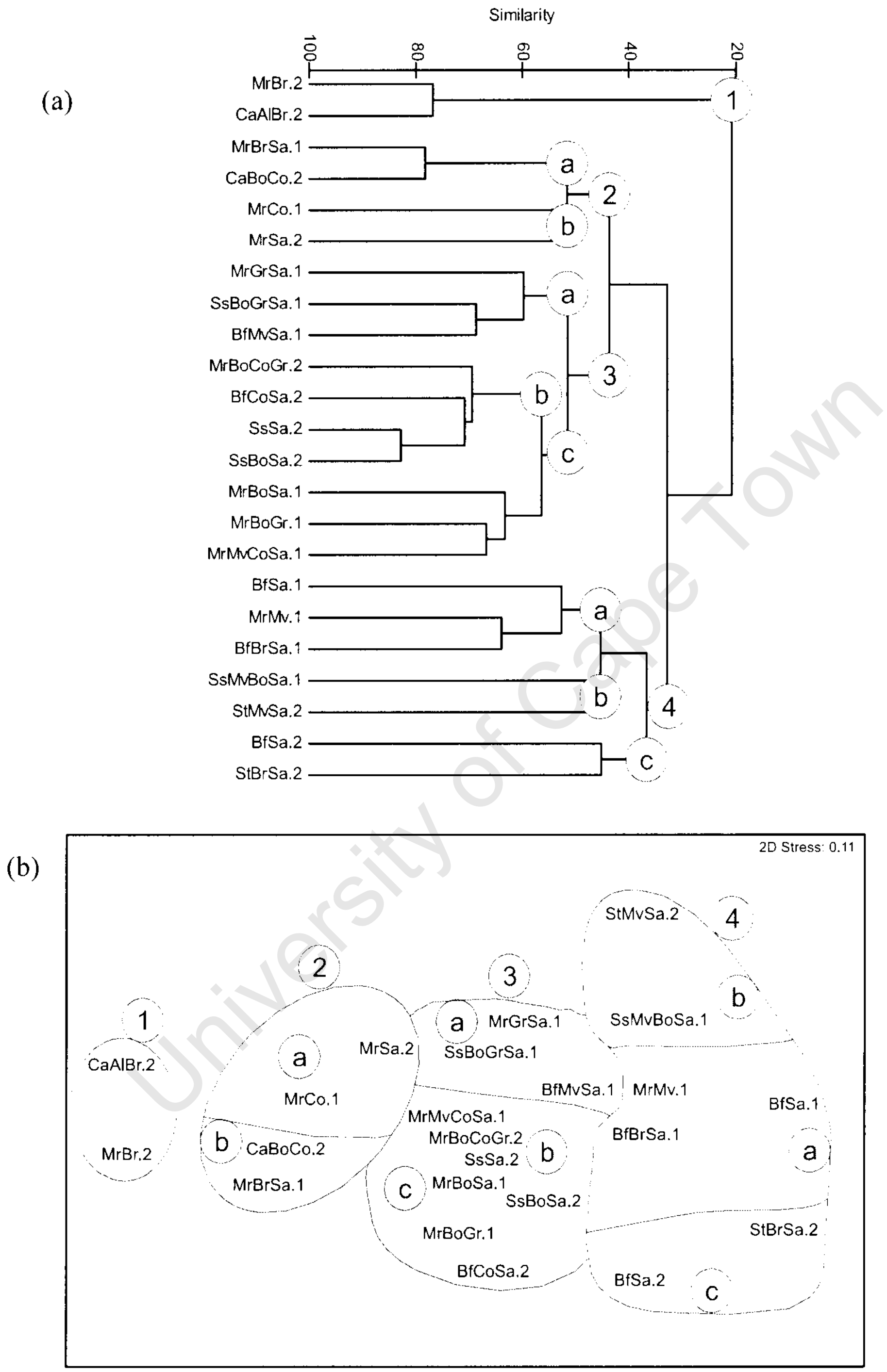


Figure 7.3 **(a) CLUSTER and (b) MDS analysis of Bray-Curtis similarity between aquatic macroinvertebrate samples from Site 2.** Sample groups are formed from samples with a high percentage similarity, based upon the community composition. Sample codes are biotope components (as per Table 4.2), followed by sample year.

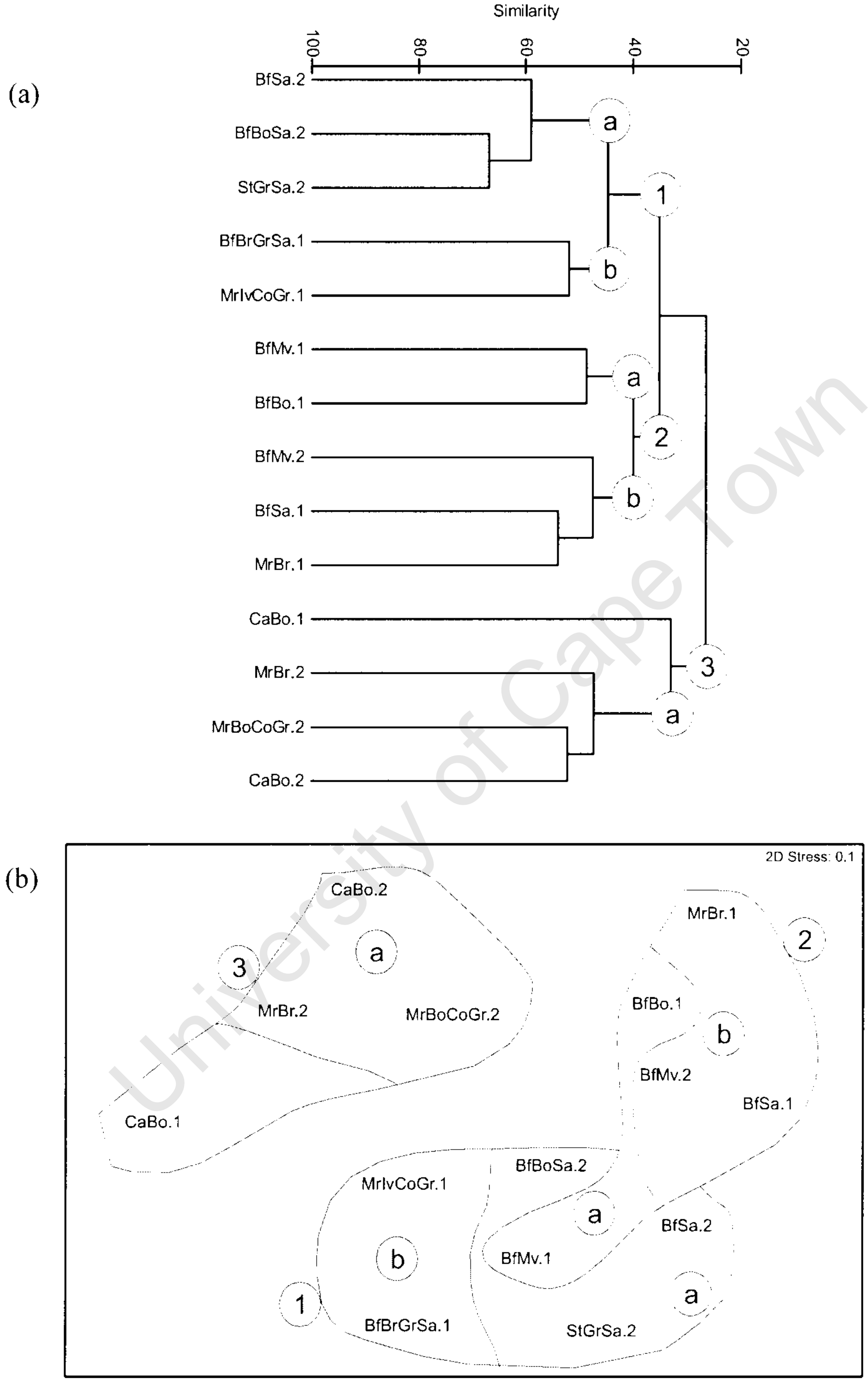


Figure 7.4 **(a) CLUSTER and (b) MDS analysis of Bray-Curtis similarity between aquatic macroinvertebrate samples from Site 3.** Sample groups are formed from samples with a high percentage similarity, based upon the community composition. Sample codes are biotope components (as per Table 4.2), followed by sample year.

Group 2b also contained a sample from a slow-flowing area with marginal vegetation. The other two samples in this group were quite different: a sample from a slow-flowing sandy area and one from a fast-flowing area over bedrock. There were no obvious similarities, based upon the coarse habitat units (flow and substratum type) or the year in which the sample was collected for group 2. Group 3, unlike the other two groups from this site, contained four samples from fast-flowing areas; one from year 1 that separated from the others collected during year two and found together in group 3a.

Based upon the separate MDS and CLUSTER analyses per site, there are some apparent similarities in the groups formed. The separation of sample groups was not clear and so it is not certain which of the factors being tested were better discriminators of aquatic macro-invertebrate community similarity, if any. The habitat descriptor codes (flow and substratum type) and the year in which the sample was collected, were further tested using the ANOSIM procedure (analysis of similarities, Clarke and Warwick 2001), a non-parametric permutation procedure that is analogous to an ANOVA (analysis of variance).

7.2.2 Testing the significance of community composition differences

The null hypothesis for these ANOSIM test statistics was that there were no justifiable differences in macroinvertebrate community composition between the sample groups being tested. In each case it was only possible to test two sample groups at a time.

Testing for differences in macroinvertebrate community composition between years and across sites

The global test of the Null hypothesis based upon year and site groups was accepted for year groups, but rejected for site groups. This means the variability in community composition between the samples collected in each of the two year groups was similar, such that the community groups could not be distinguished based upon the year in which they were collected. When samples were grouped by site however, the separation of sample groups was weak (R = 0.413, p = 0.001). However there did appear to be some site-by-site similarities worth investigating. This was done by completing the pair-wise test for each site comparison separately.

The null hypothesis was rejected when comparing Site 1 with Sites 2 and 3 respectively, and accepted between Sites 2 and 3. Thus the macroinvertebrate community from Site 1 was significantly different to those at the other two sites: Site 2, R = 0.838, p = 0.001; and Site 3; R = 0.937, p = 0.001. There were no distinguishing differences in (family level) community composition between Sites 2 and 3, and so the null hypothesis of no differences in community composition was accepted.

It appeared that there were differences between some of the site groups worth investigating further. This was done by testing differences between the habitat descriptor codes based upon the community composition of each between the three sites. The aim was to use the habitat descriptor codes, and associated habitat maps that recorded changes in available habitat between the two sampling periods, to search for changes in community composition that could be attributed to changes in the available habitat.

Testing for differences between coarse habitat units

The Global test for similarities in sample variability for all samples collected from all sites in each of the two years, based upon habitat descriptor codes (flow types, substratum types and other, as per Table 4.2) was accepted; there were no justifiable differences in community composition based upon the habitat descriptor codes. It appeared that the habitat descriptor codes chosen to represent macroinvertebrate biotopes were not able to detect significant differences in the invertebrate assemblages after two years. It may have been that there was insufficient biotope heterogeneity for the establishment of discriminant macroinvertebrate communities. An alternative explanation could have been a depauperate macroinvertebrate fauna. In order to further test for discriminant communities, the habitat descriptor codes were reduced *a posteriori* into the coarser habitat units used in the MDS and CLUSTER analyses to assess whether more meaningful habitat units could be distinguished. The flow types were reduced to two categories only: fast and slow. The substratum types were reduced to two different combinations. The first combination of categories included rock (bedrock, boulder and cobble), sand (sand and silt), rock/sand, sand/vegetation (marginal), rock/sand/vegetation, rock/vegetation and vegetation. The second combination of categories were bedrock, coarse (boulder and cobble), mixed (boulder, cobble, gravel, sand or any combination thereof), fine (gravel, sand and silt) and marginal vegetation. These sets of habitat units were tested independently and are reported upon next.

Coarse flow type

The relevance of the two different flow types (fast or slow) to macroinvertebrate community composition was tested. The null hypothesis of no differences in community composition between samples from faster flowing versus slower flowing areas was accepted for Sites 1 and 2. This means that the variability in community composition between sample groups from both these two sites, based upon fast and slow flow types, was barely perceptible. The null hypothesis was weakly rejected for Site 3; samples were separable but there was overlap between community groups ($R = 0.507$, $p = 0.002$).

Coarse substratum type 1

The relevance of the first set of substratum type combinations to macroinvertebrate community composition was tested. The global test for substratum type based upon rock (bedrock, boulder and cobble), sand (sand and silt), rock/sand, sand/vegetation (marginal), rock/sand/vegetation, rock/vegetation and vegetation did not produce any justifiable differences between community composition.

Coarse substratum type 2

The relevance of the second set of substratum type combinations to macroinvertebrate community composition was tested for all three sites. The global test of community differences being based upon substratum types such as bedrock, coarse (boulder and cobble), mixed (boulder, cobble, gravel, sand or any combination thereof), fine (gravel, sand and silt) and marginal vegetation was rejected for Site 1 ($R = 0.552$, $p = 0.030$) and accepted at both Sites 2 and 3. The separation, for samples from Site 1, was relatively weak with overlap occurring between sample groups. This means that there was some separation of invertebrate assemblages by coarse substratum type at Site 1 but not at Sites 2 and 3.

Some of the pair-wise tests for each of the three sites contained a number of substratum type combinations that revealed separable community assemblages. Each site is reported on in turn.

Site 1

The relevance of the second set of substratum type combinations to macroinvertebrate community composition at Site 1 was tested. The null hypothesis of community differences being based upon substratum types such as bedrock, and mixed and fine substratum types respectively, and also coarse and mixed substratum types, was rejected in all cases. This means the macroinvertebrate communities at Site 1 were different between different substratum types; bedrock communities were different to those on mixed ($R = 1$, $p = 0.143$) and fine ($R = 1$, $p = 0.143$) substratum types, and also to those on coarse and mixed substratum types ($R = 1$, $p = 0.333$).

Site 2

The relevance of the second set of substratum type combinations to macroinvertebrate community composition at Site 2 was tested. The null hypothesis of community differences being based upon substratum types such as bedrock, and mixed and fine substratum types respectively, and also between coarse substrata and the marginal vegetation habitat type, was rejected in all cases. This means the macroinvertebrate communities at Site 2 were different for samples between different substrata; bedrock communities were different to those over mixed ($R = 0.826$, $p = 0.001$) and fine substrata ($R = 0.929$, $p =$

0.067). This was also the case at Site 1. There was also strong separation of invertebrate communities found in coarse substratum types when compared to those found in marginal vegetation.

Site 3

The relevance of the second set of substratum type combinations to macroinvertebrate community composition at Site 3 was tested. The null hypothesis of community differences being based upon substratum types such as mixed substrata, bedrock, marginal vegetation, fine and coarse substrata, was rejected in all cases. This means the macroinvertebrate communities at Site 3 were different for different substrata; communities from mixed substrata were different to those found on bedrock ($R = 0.75$, $p = 0.0067$), marginal vegetation ($R = 0.571$, $p = 0.067$), over fine ($R = 0.536$, $p = 0.133$) and coarse substrata ($R = 0.521$, $p = 0.029$). The differences in community composition between areas of mixed substrata and those with bedrock were the strongest.

Summarising significance of ANOSIM tests

The MDS and CLUSTER analyses highlighted the potential for differences between macroinvertebrate communities at the three sites, between years and also across the sampled habitat units, to exist. ANOSIM tests were employed to test for significance between community differences. First, at the coarsest level, all samples collected during both years from all three sites revealed that the invertebrate community (at family level) found at Site 1 was very different to those at Sites 2 and 3 (Table 7.1); the invertebrate communities from the latter two sites were generally indistinguishable. Macroinvertebrate communities could not be distinguished between the two sample years. Further there were no differences between communities based upon the habitat descriptor codes (as per Table 4.2) at any of the three sites.

It was thus necessary to define coarser habitat units *a posteriori* to assess whether invertebrate communities could be separated at a coarser level. Two groupings of habitat units were tested, with the first also failing to reveal separable differences in community composition based upon the chosen habitat categories: rock (bedrock, boulder and cobble), sand (sand and silt), rock/sand, sand/vegetation (marginal), rock/sand/vegetation, rock/vegetation and vegetation. The second revision of coarse habitat units reduced the six flow types (Table 4.2) to two; fast and slow. The six substratum type categories (Table 4.2) were reduced to four: bedrock, coarse (cobble and boulder), fine (gravel and sand) and mixed (any combination of the previous categories). A further category for samples collected in marginal vegetation was added. There were justifiable differences shown in the community composition of certain comparisons between pairs of this second set of coarse habitat units (see Table 7.1).

The communities from bedrock were different to those from both fine and mixed substratum types at all three sites.

Table 7.1 Statistically significant comparisons based upon composition of aquatic macroinvertebrates in sample groups.

ANOSIM test performed	Comparison shown to be significantly different
Global site (longitudinal zone) test	Site 1 (mountain stream) vs. Sites 2 and 3 (foothill)
Global flow type test	Site 3, fast vs. slow flow type
Pair-wise substratum type tests	Site 1
	Bedrock vs. mixed
	Bedrock vs. fine
	Coarse vs. mixed
	Site 2
	Bedrock vs. mixed
	Bedrock vs. fine
	Coarse vs. marginal vegetation
	Site 3
	Bedrock vs. mixed
	Bedrock vs. fine
	Bedrock vs. marginal vegetation
	Bedrock vs. coarse

The communities of Sites 1 and 2 did not separate based upon flow type while those at Site 3 did. The communities of all three sites that were separated based upon some of the substratum type pair-wise comparisons were tested further using SIMPER. For example, the macroinvertebrates from a **bedrock** community were compared against those from a **fine** substratum type community. The results of the SIMPER analyses are reported upon next.

7.2.3 Determining discriminating taxa

All the above comparisons (Table 7.1), where differences were shown to exist, were tested further using the SIMPER 'similarity percentages' routine. This analysis determines how individual taxa contribute towards the separation of two groups of samples, or the closeness of samples within a group. Results are presented as the percentage contribution of each taxon towards the sample identity; the taxa are listed in decreasing order of such contribution.

Species dissimilarity

The average dissimilarity between two pairs of samples is represented as the sum of the individual contributions made by the two different sets of taxa. In this instance contributions for each pair being compared are measured as a taxon's particular abundance in a sample (columns 1 and 2, see Table 7.2). The descending order in which the taxa are listed and their relative contributions towards this total average dissimilarity (Av.Diss, Column 3) is the first important factor in determining which taxon may be responsible for group separation. The second element associated with this is the ratio between a taxon's average contribution and the standard deviation (SD) of that contribution across all pairs of samples making up this average (Diss/SD, Column 4). A good discriminating taxon is one with a high average abundance that contributes relatively consistently for all pairs of comparisons made, which thus has a low SD and a higher ratio (>1). Most emphasis should be placed on the order in which the taxa are displayed, namely their decreasing contribution to the between-group dissimilarity (column 3). Columns 1 and 2 also aid the interpretation by giving the average abundance for each taxon in the group.

Table 7.2 **Average dissimilarities between sample groups at Site 1 and Sites 2/3.** Group ms = mountain stream (Site 1), Group fh = foothill zone (Sites 2 and 3). Av.Abund = average abundance, Av.Diss = average dissimiliarity, Diss/SD = the average dissimilarity/standard deviation of the average, Contribut% = % contributed towards total average dissimilarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average dissimilarity.

Average dissimilarity = 82%	**1**	**2**	**3**	**4**		
	Group ms	**Group fh**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Amphipoda	35.2	3.0	33.4	2.4	40.7	40.7
Chironomidae	4.5	9.4	6.9	1.2	8.4	49.0
Baetidae	0.0	7.2	6.3	0.8	7.7	56.7
Lumbriculidae	6.3	0.5	6.0	1.6	7.3	64.0
Gomphidae	0.0	3.8	4.1	1.1	4.9	68.9
Simuliidae	0.0	5.2	3.6	0.5	4.3	73.2
Libelullidae	0.0	3.0	3.2	0.8	3.9	77.1
Planariidae-Planaria	3.3	0.0	2.9	0.9	3.5	80.6

Species similarity

In the same way, yet perhaps of less practical significance, one can examine the contribution each species makes to the average similarity within a group. The more abundant a species is within a group, the more it will contribute to the intra-group similarities. A species typifies the group if it is found at a consistent abundance throughout. This does not indicate whether the taxon concerned discriminates one group from another; it may be typical of a number of groups.

Global site test: Site 1 vs. Sites 2 and 3

The average of the Bray-Curtis dissimilarities between all pairs of sites in the **mountain stream** group and the **foothill** groups, was 82% (Table 7.2). Half of this total dissimilarity was contributed by Amphipoda, which occurred abundantly in the mountain stream (Site 1). The ratio between the average contribution and the standard deviation is relatively high indicating that this species, based upon its abundance, is a good discriminating species for Site 1. Four other families contributed between 4% and 5% to the total average dissimilarity; these were Planariidae, found only in the mountain stream, and Gomphidae, Simuliidae and Libellulidae found exclusively in the foothills.

Global flow type test Site 3: slow vs. fast flow

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **slow-flow** and the **fast-flow** groups was 70% (Table 7.3). This was contributed mainly by Chironomidae and Baetidae (both 15% toward the total average dissimilarity), with the former being three times and the latter 12 times as abundant in the fast-flow group. Although these two species were the most abundant at this site, neither were good discriminators of the fast-flow category. Chironomids and Baetids were abundant in other categories at other sites and are generally ubiquitous. No other species presented itself as a good discriminator as all the ratios of the average dissimilarity:standard deviation (Diss/SD) were low. Other abundant taxa were Libellulidae and Simuliidae, both contributing approximately 10% to the total dissimilarity, the latter being more prominent in the fast-flow habitats.

Table 7.3 **Average dissimilarities between fast and slow flow samples at Site 3.** Av.Abund = average abundance, Av.Diss = average dissimiliarity, Diss/SD = the average dissimilarity/standard deviation of the average, Contribut% = % contributed towards total average dissimilarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average dissimilarity.

Average dissimilarity = 70%	**1**	**2**	**3**	**4**		
	Group slow	**Group fast**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironomidae	6.8	21.2	10.9	1.4	15.5	15.5
Baetidae	0.8	12.1	10.8	1.1	15.4	30.9
Libelullidae	7.5	4.7	7.4	1.5	10.5	41.4
Simuliidae	0.3	10.6	6.7	0.8	9.6	51.0
Gomphidae	5.0	4.9	5.6	1.1	7.9	58.9
Athericidae	0.6	4.3	3.4	1.5	4.8	63.7
Amphipoda	2.3	3.7	2.9	1.4	4.2	67.9
Leptoceridae	2.8	1.9	2.8	1.5	3.9	71.8
Coenagrionidae	1.3	3.0	2.7	0.9	3.8	75.6
Elmidae	1.5	2.3	2.0	1.5	2.8	78.5

Since no species presented itself as a good discriminator of either the two flow groups, it was useful to examine the average similarity of taxa between fast and slow-flow groups, to compare species typically associated with these two groups at this site.

The average of the Bray-Curtis similarities between all slow-flow samples, based on abundance of taxa, was 45% (Table 7.4a). Libellulidae and Chironomidae both contributed one third towards this total and both had high ratios of Sim/SD: both may be considered typical of this flow type. This is not to say however, they are discriminant species for slow-flow as they may be shown to be typical of other sample groups as well. Leptoceridae and Gomphidae also each contributed 10% toward the total average similarity of the slow-flow group.

The fast-flow group samples had an average Bray-Curtis similarity (based on abundance) of 33% (Table 7.4b). Again Chironomidae contributed a maximal two thirds of this total average similarity, with Baetidae contributing the next largest 20%. Athericidae and Simuliidae both contributed 8 and 7% respectively.

Table 7.4 **Average similarity between fast and slow flow sample groups at Site 3.** Av.Abund = average abundance, Av.Sim = average similiarity, Sim/SD = the average similarity/standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average similarity.

(a) Slow flow

Average similarity: 45% **Taxon**	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Libelullidae	7.5	14.1	4.2	30.9	30.9
Chironomidae	6.8	13.8	2.4	30.2	61.1
Leptoceridae	2.8	5.0	1.8	11.0	72.0
Gomphidae	5.0	5.0	0.8	10.8	82.9

(b) Fast flow

Average similarity: 33% **Taxon**	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironomidae	21.2	12.5	2.5	37.3	37.3
Baetidae	12.1	7.4	1.1	21.9	59.2
Athericidae	4.3	2.8	1.1	8.4	67.6
Simuliidae	10.6	2.4	0.9	7.0	74.7
Amphipoda	3.7	2.0	0.9	5.9	80.6

Thus, the main difference at this site between the fast and slow-flow group was the abundant presence of Libellulidae in the slow-flow category (Table 7.4a) and the abundant presence of Baetidae in the fast-flow category (Table 7.4b), in spite of their respectively low Diss/SD ratios (Table 7.3).

Pair-wise <u>substratum type</u> comparisons: Sites 1 to 3

<u>Site 1</u>

Three pair-wise comparisons were shown to contain differences based upon the substratum type of the samples in the two groups (Table 7.5) and each is presented in turn.

Table 7.5 **Average dissimilarity of substratum type sample groups at Site 1.** Av.Abund = average abundance, Av.Diss = average dissimiliarity, Diss/SD = the average dissimilarity/standard deviation of the average, Contribut% = % contributed towards total average dissimilarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average dissimilarity.

(a) Bedrock vs. mixed

Average dissimilarity = 62%	**Group mixed**	**Group bedrock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Amphipoda	30.7	8.1	34.4	5.0	55.2	55.2
Lumbriculidae	5.7	1.4	7.2	2.2	11.5	66.8
Chironomidae	5.8	2.0	6.8	0.9	11.0	77.7
Planariidae-Planaria	2.9	0.0	4.1	1.0	6.6	84.3
Elmidae	2.7	0.0	4.0	4.2	6.4	90.7

(b) Bedrock vs. fine

Average dissimilarity = 64%	**Group fine**	**Group bedrock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Amphipoda	42.1	8.1	43.2	31.5	67.5	67.5
Lumbriculidae	7.8	1.4	6.5	1.1	10.2	77.7
Naididae	2.7	1.4	3.3	7.5	5.1	82.8
Elmidae	3.0	0.0	3.2	1.3	5.0	87.8
Chironomidae	3.1	2.0	2.2	2.9	3.4	91.1

(c) Coarse vs. mixed

Average dissimilarity = 53%	**Group coarse**	**Group mixed**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Amphipoda	75.0	30.7	23.8	5.2	44.6	44.6
Elmidae (A)	13.0	0.0	6.9	16.7	13.0	57.7
Elmidae	11.5	2.7	4.7	5.9	8.8	66.5
Planariidae-Planaria	11.5	2.9	4.6	2.5	8.7	75.2
Lumbriculidae	11.2	5.7	2.9	1.6	5.5	80.7
Chironomidae	2.5	5.8	2.3	0.8	4.4	85.1
Hydraenidae (A)	3.5	0.0	1.9	16.7	3.6	88.7
Notonemouridae	3.5	0.2	1.8	6.4	3.4	92.0

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **mixed** substratum type groups was 62%. This was due to the abundance of Amphipoda in the mixed category that contributed more than half of the total dissimilarity (Table 7.5a). This taxon at Site 1 had a high Diss/SD ratio, indicating that it could be considered a good discriminating species for the mixed substratum type at this site. The next two largest contributions to the total average dissimilarity were

11% from the Lumbriculidae and Chironomidae, yet neither were particularly good discriminators. Of the two families found in the mixed group, only Elmidae (type A) had a relatively high Diss/SD ratio and thus may also be considered to be a discriminating taxon for the mixed substratum category.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **fine** substratum type groups was 64% (Table 7.5b). Again this was attributed to an abundance of Amphipoda in the fine category, contributing two thirds to the total dissimilarity (Table 7.5b). This taxon had an extremely high Diss/SD ratio at Site 1, indicating it could be considered a good discriminating species for the fine substratum type. As above, Lumbriculidae contributed 10% towards the total dissimilarity. Naididae, which was not particularly abundant, had a relatively high Diss/SD ratio and thus was also considered to be a discriminating species for the fine substratum type group.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **coarse** and **mixed** substratum type groups was 53% (Table 7.5c). Again this was attributed to the Amphipoda, which were twice as abundant in the coarse vs. the mixed category, and contributed 45 % toward the total dissimilarity. The next largest contributors were Elmidae (A) with 13%, and Elmidae and Planariidae both with 9%. Elmidae (A), along with Hydraenidae (A), which contributed only 4% towards the total average dissimilarity, were considered the best discriminating taxa for the coarse group as they had the highest Diss/SD ratios.

The fact that Amphipoda were considered discriminators for each of the three substratum type comparisons suggests it can exist on several different kinds of substrata, thus negating its usefulness as a discriminator of substratum types. Overwhelmingly, the prominence of the Amphipoda in the site (zonal) comparison was the single largest factor separating Site 1 (in the mountain stream) from Sites 2 and 3 (in the foothill zone).

Site 2

Three pair-wise comparisons were shown to contain differences based upon the substratum type of the samples in the two groups (Table 7.6) and each is presented in turn.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **mixed** substratum type groups was 80% (Table 7.6a). The largest contributor towards this was Simuliidae, which was dominant in the bedrock category, and responsible for 47% of the total average dissimilarity. Simuliidae also had the highest Diss/SD ratio and thus was the best discriminator for the bedrock group.

The next two most abundant taxa in the bedrock category were Baetidae and Chironomidae, being responsible for 24% and 12% of the total average dissimiliarity respectively and both with relatively high diss/SD ratios thus representing good discriminating species for bedrock.

Table 7.6 **Average dissimilarity between substratum type sample groups at Site 2.** Av.Abund = average abundance, Av.Diss = average dissimiliarity, Diss/SD = the average dissimilarity/standard deviation of the average, Contribut% = % contributed towards total average dissimilarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average dissimilarity.

(a) Bedrock vs. mixed

Average dissimilarity = 80% **Taxon**	**Group mixed** Av.Abund	**Group bedrock** Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Simuliidae	1.0	48.2	35.9	7.2	44.7	44.7
Baetidae	4.5	28.9	19.4	3.1	24.2	68.9
Chironomidae	5.9	17.3	9.3	2.2	11.6	80.5
Elmidae	1.5	6.5	4.1	1.9	5.1	85.7

(b) Bedrock vs. fine

Average dissimilarity = 86% **Taxon**	**Group fine** Av.Abund	**Group bedrock** Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Simuliidae	0.0	48.2	39.0	7.1	44.9	44.9
Baetidae	2.1	28.9	21.4	3.4	24.6	69.5
Chironomidae	4.4	17.3	11.2	2.5	12.9	82.4
Elmidae	0.6	6.5	5.1	2.2	5.8	88.2

(c) Coarse vs .marginal vegetation

Average dissimilarity = 77% **Taxon**	**Group coarse** Av.Abund	**Group m. veg.** Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Amphipoda	22.2	0.0	17.5	0.8	22.7	22.7
Baetidae	11.4	2.2	14.0	1.0	18.2	40.8
Chironomidae	8.1	4.0	8.0	1.2	10.4	51.2
Elmidae	5.8	0.0	6.2	1.6	8.1	59.3
Gomphidae	3.0	1.0	4.0	1.2	5.1	64.4
Leptoceridae	1.6	0.7	3.1	0.7	4.0	68.4
Simuliidae	1.8	0.0	2.7	0.5	3.5	71.9
Hydroptilidae	0.0	1.7	2.5	2.0	3.2	75.1
Lumbriculidae	3.2	0.0	2.4	0.7	3.1	78.2

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **fine** substratum type groups was 86% (Table 7.6b). As above, this was attributed to Simuliidae, which contributed half of the total average dissimilarity and again was shown to the best discriminator for bedrock when compared with the fine substratum type group. Again, as above, the two next most abundant taxa in the bedrock category were Baetidae and Chironomidae, with 25% and 13% respectively.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **coarse** and **marginal vegetation** groups was 77% (Table 7.6c). Amphipoda and Baetidae were the largest contributors towards this dissimilarity, both being more abundant in the coarse group and contributing 23 and 18% respectively. These were followed by Chironomidae (10%) and Elmidae (A, 8%). No particular taxon was shown to be a good discriminator of any of the two groups being compared. This was probably due to there being only one marginal vegetation sample in this group; SIMPER requires a minimum of two samples to generate an appropriate statistical comparison.

Site 3

Four pair-wise comparisons were shown to contain differences based upon the substratum type of the samples in the two groups (Table 7.7) and each is presented in turn. For all pair-wise comparisons there were too few replicates of the bedrock category for meaningful standard deviations to be calculated, thus ratios of Diss/SD are not particularly useful as they are based upon insufficient data. Nonetheless, the results of the taxon contributions towards the total average dissimilarity are presented for each of the four cases.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **mixed** substratum type groups was 72% (Table 7.7a). The largest contributors towards this were Baetidae with 17% and dominant in the bedrock category, followed by Amphipoda and Libelullidae both with 13% and dominant in the mixed substratum category. Chironomidae and Gomphidae were the next most abundant with 12% and 11% respectively contributed toward the total average dissimilarity. No taxon presented itself as a particularly good discriminator for either of these two substratum categories. The highest ratio of Diss/SD was for Leptoceridae, found in the mixed group. Athericidae, not found at any of the other sites, occurs in the SIMPER outputs for all of the four pair-wise comparisons presented below.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **fine** substratum type groups was 65% (Table 7.7b). Again, the same group of families contributed towards this in somewhat different proportions to that already presented for this site: Baetidae (30%), Libellulidae (12.5%), Chironomidae (12%) and Gomphidae (11%). Again no particular taxon presented itself as a good discriminator in this group.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **marginal vegetation** groups was 71% (Table 7.7c). Again Baetidae, which was most abundant in the bedrock samples, contributed the largest portion (24%) towards the total average dissimilarity followed by Chironomidae (14%), Libellulidae (10%), Coenagrionidae (8%) and Leptoceridae (7.5%).

Table 7.7 **Average dissimilarity between substratum type sample groups at Site 3.** Av.Abund = average abundance, Av.Diss = average dissimiliarity, Diss/SD = the average dissimilarity/standard deviation of the average, Contribut% = % contributed towards total average dissimilarity per taxon, and Cum.% = cumulative percentage total of all taxa towards average dissimilarity.

(a) Bedrock vs. mixed

Average dissimilarity = 72%	**Group mixed**	**Group bedock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Baetidae	1.7	19.2	12.5	1.2	17.3	17.3
Amphipoda	10.6	0.9	9.7	0.8	13.4	30.6
Libelullidae	11.1	1.1	9.2	1.3	12.6	43.3
Chironomidae	15.9	11.4	9.0	1.3	12.4	55.7
Gomphidae	10.0	2.3	7.7	1.3	10.6	66.3
Athericidae	2.5	4.6	3.2	1.5	4.4	70.7
Leptoceridae	3.9	0.5	2.9	1.8	4.0	74.7
Coenagrionidae	3.2	0.0	2.2	0.7	3.1	77.7
Ceratopogonidae	2.5	0.7	2.0	1.1	2.8	80.5

(b) Bedrock vs. fine

Average dissimilarity = 65%	**Group fine**	**Group bedrock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Baetidae	0.5	19.2	19.6	1.2	30.0	30.0
Libelullidae	5.7	1.1	8.1	1.5	12.5	42.5
Chironomidae	3.4	11.4	7.9	1.0	12.1	54.6
Gomphidae	5.6	2.3	7.2	1.1	11.0	65.6
Athericidae	0.4	4.6	4.2	1.0	6.4	72.0
Notonemouridae	1.9	0.0	2.8	0.7	4.2	76.2
Elmidae	2.0	1.8	2.2	1.2	3.4	79.6

(c) Bedrock vs. marginal vegetation

Average dissimilarity = 71%	**Group m. veg.**	**Group bedrock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Baetidae	0.7	19.2	17.2	1.1	24.2	24.2
Chironomidae	8.5	11.4	9.9	1.5	13.9	38.1
Libelullidae	5.6	1.1	6.9	1.6	9.8	47.8
Coenagrionidae	4.0	0.0	5.7	1.0	8.0	55.8
Leptoceridae	4.2	0.5	5.3	2.7	7.5	63.3
Athericidae	0.9	4.6	4.3	1.6	6.1	69.3
Amphipoda	2.8	0.9	3.3	1.3	4.6	73.9
Simuliidae	0.0	2.1	2.4	3.1	3.3	77.3
Elmidae	0.0	1.8	2.1	4.1	3.0	80.3

(d) Bedrock vs. coarse

Average dissimilarity = 70 %	**Group coarse**	**Group bedrock**				
Taxon	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Baetidae	8.2	19.2	14.6	1.5	20.6	20.6
Chironomidae	17.9	11.4	11.8	1.5	16.7	37.3
Amphipoda	17.0	0.9	10.4	0.6	14.7	51.9
Simuliidae	13.0	2.1	7.6	0.8	10.8	62.7
Athericidae	2.0	4.6	3.7	1.3	5.3	68.0
Elmidae	3.6	1.8	2.6	1.4	3.6	71.6
Lumbriculidae	3.1	0.0	2.4	0.9	3.4	74.9
Libelullidae	1.1	1.1	1.9	0.6	2.6	77.5

The latter three were more abundant in the marginal vegetation group, with Leptoceridae again presenting the highest Diss/SD ratio for this category. Simuliidae and Elmidae, both with low contributions towards the total average dissimilarity, were found exclusively in the bedrock group and both had high Diss/SD ratios. These two families may thus be considered good discriminators of bedrock when compared to marginal vegetation.

The average of the Bray-Curtis dissimilarities between all pairs of samples in the **bedrock** and **coarse** substratum type groups was 70% (Table 7.7d). Again, the same group of families contributed towards this in somewhat different proportions to that already presented for this site: Baetidae (21%) and more abundant in the bedrock category; Chironomidae (17%), Amphipoda (15%) and Simuliidae (11%), all more abundant in the coarse category; while some of the others already mentioned above were smaller contributors. Again no particular taxon presented itself as a good discriminator in this group.

Summarising discriminating taxa from the SIMPER analyses

The test for discriminating families between sites, based upon average dissimilarity (Table 7.2), showed the Amphipoda to be the dominant and discriminating species responsible for the separation of Site 1 from Sites 2 and 3 (Figure 7.1). Fast and slow-flow types were compared, based upon the average dissimilarity at Site 3. Iit was shown that the Chironomidae and the Baetidae, both of which were more abundant in the fast-flow samples, were responsible for the separation of fast from slow-flow type samples, yet neither was shown to be a particularly good discriminator for the fast-flow type. Since no taxon presented itself as a discriminator, the average similarity results were used to elucidate further. The presence of the Libellulidae in the slow-flow category of samples and the abundance of the Baetidae and Chironomidae in the fast-flow category of samples were shown to be the main differences between the two flow categories (Tables 7.4a and 7.4b), even with their low Diss/SD ratios. These two analyses were followed by a range of pair-wise comparisons between the coarse habitat units (as per Table 7.1) shown to contain significant differences on a site by site basis. At each of the three sites (Table 7.8) bedrock was compared to mixed and then to fine respectively. Further comparisons (Table 7.9) were coarse vs. mixed substrata at Site 1; coarse substrata vs. marginal vegetation habitat at Site 2; and bedrock substratum type vs. marginal vegetation and bedrock vs. coarse substrata at Site 3.

The Amphipoda were shown to be the overwhelming discriminating taxon for the mixed substrata at Site 1 (Table 7.8). This same taxon was the dominant taxon present so not surprisingly it was shown to be a discriminating species for fine substrata as well (Table 7.8). The Simuliidae, Baetidae and Chironomidae were shown to be discriminant taxa for bedrock substrata at Sites 2 and 3, when compared against both mixed and fine substrata (Table 7.8). The Chironomidae appeared to be either more abundant on bedrock or equally abundant between the two habitat categories being compared (Tables 7.8 and 7.9). These SIMPER results were not conclusive across sites, perhaps due to the nature of the analysis only being able to take into account two sample groups at a time. It was thus necessary to do further analyses to determine which families were dominant between year groups and across sites so that a trajectory of recovery may be investigated.

Table 7.8 **Discriminating taxa for significantly different coarse-habitat units found at all three sites.** Brackets are the percentage that each taxon contributes towards the total dissimilarity between each pair-wise comparison. Italicised taxa are those where the Diss/SD ratio was low; these species were abundant yet are not discriminating species for the particular coarse habitat unit.

	BEDROCK	MIXED
Site 1	Elmidae (6%)	Amphipoda (55%)
Site 2	Simuliidae (48%) Baetidae (24%) Chironomidae (12%)	
Site 3	*Baetidae (17%)* *Chironomidae (12%)*	*Amphipoda (13%)* *Libellulidae (13%)* *Gomphidae (11%)* Leptoceridae (3%) Athericidae (4%)
	BEDROCK	FINE
Site 1	Chironomidae (3%)	Amphipoda (68%) Naididae (5%)
Site 2	Simuliidae (48%) Baetidae (24%) Chironomidae (13%)	Elmidae (6%)
Site 3	*Baetidae (30%)* *Chironomidae (12%)* *Athericidae (6%)*	*Libellulidae (13%)* *Gomphidae (11%)*

The first of these additional analyses compared the dominant taxa in each sample group to a reference condition compiled for upland Western Cape rivers by Dallas and Day (2007). The second used the principles behind SASS scoring to generate pseudo-SASS scores between the two year groups at each site (see Section 2.4). These two analyses were used to assess recovery over the year interval between sites. Lastly, the Present Ecological Status (Brown *et al.* 2001) was calculated for the mountain stream (Site 1) and for the foothills (Sites 2 and 3) as a way of determining the impacts upon the quality of aquatic habitat available to the organisms.

Table 7.9 **Discriminating taxa for significantly different coarse-habitat units found at one of the sites.** Brackets are the percentage each taxon contributes towards the total dissimilarity between each pair-wise comparison. Italicised taxa are those where the Diss/SD ratio was low; these species are abundant yet are not discriminating species for the particular coarse habitat unit.

	BEDROCK	MARGINAL VEGETATION
Site 3	*Baetidae (24%)*	*Libellulidae (10%)*
	Chironomidae (14%)	
	Simuliidae (2%)	*Coenagrionidae (8%)*
	Elmidae (2%)	Leptoceridae (8%)
	BEDROCK	COARSE
Site 3	*Baetidae (21%)*	*Amphipoda (15%)*
	Chironomidae (17%)	
		Simuliidae (11%)
	MIXED	COARSE
Site 1		Amphipoda (45%)
		Elmidae (A) (13%)
		Elmidae (6%)
		Planariidae (9%)
		Hydraenidae (A)
		Notonemouridae (2%)
	MARGINAL VEGETATION	COARSE
Site 2		*Amphipoda (23%)*
		Baetidae (18%)
		Chironomidae (10%)
		Elmidae (8%)

7.2.4 Other measures of recovery

Comparing recovery data to reference condition data

Dallas and Day (2007) developed an aquatic macro-invertebrate reference condition for upland (mountain stream and foothill) sites in the Western Cape, together with a list of expected SASS taxa. The list includes details pertaining to seasonality and biotope preferences. This database was useful as it takes into account the intrinsic variability amongst reference conditions. Ranges were set for five categories of reference condition, from 'richer than reference', a biological hotspot, down to 'well below reference' and 'impoverished'. Frequency tables were calculated for those organisms expected to occur at reference sites and also for those expected to increase in response to disturbance.

Eight of the 20 taxa expected to occur at reference upland sites were absent from the Silvermine sites (Table 7.10). These were the Heptageniidae, Leptophlebiidae, Teloganodidae, Corydalidae, Ecnomidae, Philopotamidae, Case-caddis 3 types and Blephariceridae. Three taxa were found at all sites in both years; Elmidae, Chironomidae and Amphipoda. Of these the Elmidae were equivalent to, or slightly more abundant, than expected except at Site 3, where the numbers were lower in the first year and very much lower in the second year. The Chironomidae were slightly more frequent than expected at each of the three sites during both years. The Amphipoda were the most surprising in that they were over-abundant, totalling approximately 80% of the organisms collected at each site, when compared to the expected frequency of 18%. When present, the following organisms occurred at lower or much lower frequencies than the expected values; Notonemouridae, Dryopidae, Helodidae, Hydraenidae, Limnichidae, Simuliidae and Tipulidae. The Baetidae were the only organisms (other than Amphipoda) that occurred at consistently higher abundances than expected, while the Athericidae were at lower abundances than expected at Site 2 and higher at Site 3.

Table 7.10 **Comparison between reference taxa listed for upland sites on Western Cape rivers (Dallas and Day 2007) and those found at all sites in both years on the Silvermine River.** Data are frequency of occurrence as a percentage based upon abundance data. ≡ = equivalent to expected frequency; ↑ or ↓ = value differs up to 20%; ↑↑ or ↓↓ = value differs between 20% and 50%; and ↑↑↑ or ↓↓↓ = value differs by greater than 50%, of the expected values.

Reference condition			Site					
Order	SASS-taxon	%	1.1	1.2	2.1	2.2	3.1	3.2
Plecoptera	Notonemouridae	68	≡	↓↓	↓↓↓			
Ephemeroptera	Baetidae > 2 types	53			↑	↑	↑	↑↑
	Heptageniidae	43						
	Leptophlebiidae	95						
	Teloganodidae	81						
Coleoptera	Elmidae	75	≡	↑	≡	≡	↓	↓↓↓
	Dryopidae	75	↓↓					↓↓↓
	Helodidae	62	↓	↓	↓↓↓	↓↓↓		
	Hydraenidae	48	↓					↓↓
	Limnichidae	28			↓↓	↓↓		↑↑
Megaloptera	Corydalidae	75						
Trichoptera	Ecnomidae	32						
	Philopotamidae	48						
	Case Caddis 3 types	48						
Diptera	Athericidae	57			↓	↓↓	↑	↑
	Blephariceridae	30						
	Chironomidae	90	↑	↑	↑	↑	↑	↑
	Simuliidae	97			↓	↓↓↓	↓↓	↓↓
	Tipulidae	33		↓	↓		↓	↓
Crustacea	Amphipoda	18	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑

The presence of 12 of the 20 reference taxa in at least one year and at least one site suggests that substantial recovery should still take place. Of the eight missing taxa, six occur in fast-flowing, clean rivers and browse on algae, periphyton and detritus found on rocks, submerged wood or aquatic vegetation (Gerber and Gabriel 2002; de Moor *et al.* 2003; Day *et al.* 2003). It is suspected that the disturbed nature of the river's substrata (Section 5.2) is retarding recovery. Presently the substrata are not well sorted according to size, and the substrata present are highly embedded due to an excess of fine sediment in the channel. Thus, there is low physical diversity and few habitats available to the macroinvertebrates. In time, if the riverine habitats were to move towards a more natural state, there would be increased sorting of different substratum size classes and a decrease in fine sediment. This in turn should lead to an increase in the biodiversity of macroinvertebrates found in the river.

Comparing all sites over two years using pseudo-SASS scores

SASS5 (Dickens and Graham 2002), used for river health assessments nationally (as at April 2008), was not in use at the time the data for this thesis were collected. The strict SASS sampling protocol was not followed for this thesis, hence there can be no direct comparison between the data sets generated here and others that have used SASS (Fowler and Harding 2000, Dallas and Day 2007 and Snaddon 2007). Nonetheless, the principle of ranking organisms' sensitivities to pollution, which is inherent in the SASS methodology, was used to generate pseudo-SASS scores for the data set collected for this thesis for inter-annual comparisons (see Section 4.3).

The three sites fared differently in the pseudo-SASS scores and average score per taxon (ASPT) between year 1 and year 2 (Table 7.11).

Table 7.11 **Pseudo-SASS scores for each site in both years.** Score = pseudo-SASS score. ASPT = average score per taxon. SIC = stones in current. SOOC = stones out of current. MV = marginal vegetation. SA = sand/silt.

	2001			2002		
	# taxa	score	ASPT	# taxa	score	ASPT
			Site 1			
SIC	3	22	7.3	3	22	7.3
SOOC	2	17	8.5	4	23	5.8
MV	4	23	5.8	5	31	6.2
SA	4	23	5.8	3	23	7.7
			Site 2			
SIC	11	71	6.5	9	83	9.2
SOOC	9	55	6.1	9	60	6.7
MV	10	10	1.0	2	10	5.0
SA	7	39	5.6	5	65	13.0
			Site 3			
SIC	7	44	6.3	5	63	12.6
SOOC	5	20	4.0	4	41	10.3
MV	7	51	7.3	7	79	11.3
SA	5	34	6.8	2	43	11.5

The number of taxa remained constant at Site 1 between years. Site 1 had the lowest scores compared to the other two sites and there was little difference between the scores of Site 1 between years. The pseudo-SASS score for SOOC and MV increased slightly in year two. The ASPT remained relatively constant for SIC and MV, was slightly lower for SOOC and higher for SA. At Site 2, the number of taxa remained relatively constant between years, other than for MV where fewer organisms were present in year 2. Generally the pseudo-SASS scores were much higher than at Site 1 and were the same or higher in year 2. The ASPT increased in all four biotope categories in year 2; the highest increase recorded for SA from 5.6 to 13. At Site 3, there were the same number or slightly fewer organisms in the four categories of samples collected in year 2 when compared to year 1. Overall, this site outperformed the other two in that the highest increases in pseudo-SASS scores were seen from year 1 to 2. Similarly ASPT increased greatly over the two-year period.

The proportions of sensitive and tolerant organisms present during both years at all three sites was calculated using the SASS4 sensitivity ratings (Thirion *et al.* 1995). Those organisms that scored between 1-7 were ranked tolerant, while those from 8-15 were ranked sensitive (Table 7.12). The frequencies were calculated in two different ways, first on the basis of abundance (numbers of organism present in each sample) and then by presence/absence, which reduces the effect of an over-dominant organism, such as the Amphipoda at Site 1. The use of the abundance figures presents a somewhat skewed picture.

Table 7.12 **The percentage of sensitive and tolerant organisms at all sites in both years.** Data are based upon organisms most abundant and upon presence/absence.

	Year 1		Year 2	
	% Sensitive	% Tolerant	% Sensitive	% Tolerant
<u>Site 1</u>				
Abundance	87	13	95	5
Presence/Absence	50	50	25	75
<u>Site 2</u>				
Abundance	8	92	6	94
Presence/Absence	38	62	35	65
<u>Site 3</u>				
Abundance	3	97	8	92
Presence/Absence	23	77	44	56

At Site 1, the presence of the Amphipoda causes the proportion of sensitive organisms to far outnumber those of the tolerant organisms, although the over-dominance by the Amphipoda here is somewhat curious in itself. At Site 2, the presence of the Baetidae in the first year and then the combined presence of Simuliidae and Baetidae in the second year were responsible for the inflated proportion of tolerant organisms. At Site 3, the presences of Chironomidae and Simuliidae in the first year, and then the Baetidae and Chironomidae in the second year were responsible for the large proportion of tolerant organisms. The general trend was an increased proportion of tolerant organisms from year 1 to 2 at Sites

1 and 2, while at Site 3 there were more sensitive taxa after two years, when compared to the first. We would expect the proportion of sensitive organisms to increase if the river was moving towards a more natural state. It appears that after two years the conditions in the river, in terms of an index based on water quality but also reflecting physical habitat, were generally worse in the second year.

Present Ecological State (PES)

An assessment of PES takes into account the number and severity of anthropogenic disturbances in river catchments. A river is classed according to an overall score (Table 4.7) indicating the potential for degradation caused by perturbations in the catchment (see Section 4.3).

Site 1 in the mountain stream scored poorly (class C). This was due to the presence of the reservoir and the abstraction of water by the Westlake Golf Course (Reinecke *et al.* 2003). Further downstream in the foothills at Sites 2 and 3, the presence of alien vegetation, notably *Acacia saligna* (Port Jackson willow), *Populus canescens* (grey poplar) and *Pennisetum clandestinum* (kikuyu) impacted negatively on the riparian score and status.

At the two foothill sites, the impact of the diversion and the reservoir are lessened. As a result the In-stream status and score is much improved (class B), while the riparian score drops lower due to the alien vegetation. Accepting that the dam will remain, the PES scores of all three sites would best be improved by halting diversion of water from the river into the weir during the dry season and even perhaps releasing water from the dam for downstream river maintenance. The continual clearing of alien vegetation and subsequent re-growth of indigenous vegetation will eventually improve the PES of Sites 2 and 3.

Table 7.13 **Present Ecological Status scores and classes for all three sites.** Site locations as per Figure 3.3.

Category	Mountain stream (Site 1)	Foothill (Sites 2 and 3)
In-stream status score	67.3	82.6
Riparian status score	73.8	70.3
In-stream status class	C	B
Riparian status class	C	C

Since the PES assessment does not take into account physical habitat heterogeneity, it offers a rather optimistic class B score for the foothill sites. The analysis of pseudo-SASS scores and the sensitive versus tolerant macroinvertebrate taxa both revealed the mountain stream to be in better condition than the foothill, based upon the presence of more sensitive organisms in the mountain stream.

Comparing recovery over four years: 2001 to 2004

The macroinvertebrate data for this study were collected over two successive summer periods; 2001 and 2002. Two subsequent B.Sc (Hons.) theses (Shelton 2003 and Mann 2004) followed the same methods and re-sampled aquatic macroinvertebrates in 2003 and 2004 at Site 2. The results of these projects are summarised in the discussion (Section 7.3.2), since the focus of their work was to compare habitat availability to the presence of the invertebrates. Their approach was slightly different to the one used here viz. comparison against a reference condition data base. In addition, the data from Shelton (2003) and Mann (2004) have been manipulated in this thesis to generate similar comparisons against the reference condition. First, the additional two years' data set were compared to the taxa expected in reference condition upland Western Cape rivers (Dallas and Day 2007). The percentages of sensitive and tolerant organisms were calculated and compared over the four years. No SASS scores were generated, since it is not certain that the biotope descriptions and the sampling techniques were the same as those used for this thesis.

The frequencies of the reference taxa fluctuated through the four-year period (Table 7.14). Of the 20 taxa described by Dallas and Day (2007), six were not found at all. These were the Heptageniidae, Corydalidae, Philopotamidae and 3 types of Case Caddis (only one type, the Leptoceridae, was present during each year) and Dryopidae. The latter taxon was present during the first two years, while the prior four were absent. The missing taxa are to be found in faster-flowing rivers on rocky or submerged woody debris. Three new families, not found in the first two years, were now present. These were Leptophlebidae, Ecnomidae and Blephariceridae. Of these, the Leptophlebidae and the Blepharicidae both occur in fast-flowing rivers on stones and are sensitive to disturbances. The Ecnomidae occur in quiet pools in slower-flow on stones, or submerged vegetation. The occurrence of these three families must indicate that there has been some improvement in the conditions and kinds of biotopes present after four years. Blephariceridae are particularly sensitive and generally only found in fast flowing pristine rivers. The frequencies of the Notonemouridae and Baetidae, which were higher or lower than the expected values, or absent during the first three years, had approached the expected values by the fourth year. The Elmidae, which were present at the expected frequency during the first two years increased steadily over the subsequent two years, while the frequency of Chironomidae remained the same. Chironomids and baetids are opportunistic and multi-voltine (completing more than one life cycle per year) thus would recover rapidly after a disturbance.

Table 7.14 **Comparison between reference taxa listed for upland sites on Western Cape rivers (Dallas and Day 2007) and those found at Site 2 over four years on the Silvermine River.** Data are frequency of occurrence as a percentage based upon abundance data. ≡ = equivalent to expected frequency; ↑ or ↓ = value differs up to 20%; ↑↑ or ↓↓ = value differs between 20% and 50%; and ↑↑↑ or ↓↓↓ = value differs by greater than 50%, of the expected values.

Reference condition			Year			
Order	SASS-taxon	%	2001	2002	2003	2004
Plecoptera	Notonemouridae	68	↓↓↓		↑	≡
Ephemeroptera	Baetidae > 2 types	53	↑	↑	↑↑↑	↑
	Heptageniidae	43				
	Leptophlebiidae	95			↓↓↓	
	Teloganodidae	81				
Coleoptera	Elmidae	75	≡	≡	↑	↑↑
	Dryopidae	75				
	Helodidae	62	↓↓↓	↓↓↓		
	Hydraenidae	48			↓↓	↑
	Limnichidae	28	↓↓	↓↓		
Megaloptera	Corydalidae	75				
Trichoptera	Ecnomidae	32				↓↓
	Philopotamidae	48				
	Case Caddis 3 types	48				
Diptera	Athericidae	57	↓	↓↓		↓↓
	Blephariceridae	30				↓↓
	Chironomidae	90	↑	↑	↑	↑
	Simuliidae	97	↓	↓↓↓	↓↓↓	↓↓
	Tipulidae	33	↓			
Crustacea	Amphipoda	18	↑↑↑	↑↑↑	↑↑	↑↑

As sediment continues to deposit in the foothills from the high sandy cliffs on the left bank (see Section 5.2), uni-voltine species do not have the same ability to re-colonise the continually disturbed biotopes, as multi-voltine. Elmidae are slow moving and occur on stones or other solid substrata (Gerber and Gabriel 2002) feeding on all manner of vegetative material (Stals and de Moor 2007). The increase of this taxon must indicate an increase in the proportion of stony substrata available, presumably as excess sediment from the sandy bank is flushed downstream. Taxa that decreased but were still present after four years included the Athericidae, Simuliidae and Amphipoda. The Helodidae, Limnichidae and Tipulidae were present during the first two years but not in years three and four. Finally the Blephariceridae and Ecnomidae were new taxa that appeared during the fourth year.

Comparing the relative proportions of sensitive and tolerant macroinvertebrates present at a particular time is a useful method of assessing whether or not aquatic river health may be improving (Table 7.15). The frequencies between the two years fluctuated slightly but did not differ particularly between years. Since there is no steadily increasing trend of sensitive organisms that would generally be found at undisturbed sites, it is concluded that the condition of the river remains generally disturbed. Any new appearances, or increases in certain of the reference taxa were countered by losses or decreases of others. The numbers of reference taxa present (Table 7.14) and the presence of tolerant and sensitive organisms (Table 7.15) remained the same over the four year period.

Table 7.15 **The percentage of sensitive and tolerant organisms at Site 2 over four years.** Data are based upon organisms most abundant and then upon presence/absence.

	% Sensitive	% Tolerant
2001		
Abundance	8	92
Presence/Absence	38	62
2002		
Abundance	6	94
Presence/Absence	35	65
2003		
Abundance	7	93
Presence/Absence	39	61
2004		
Abundance	7	93
Presence/Absence	41	59

7.3 Discussion

This chapter has dealt with the use of aquatic macroinvertebrates as indicators of river health and as a means to assess the trajectory of recovery of a riverine ecosystem. The methods applied compared the state of the river at annual intervals over four years to a reference condition that was assumed to have existed prior to the fire. The discussion is separated into three main sections. The first deals with insights gleaned from the analysis of data collected during the first two years for this thesis. Relevant data and discussion on general management and restoration (Fowler and Harding 2000; Harding 2000) and pertinent to biotope and community analyses (Reinecke and King 2003; Reinecke *et al.* 2003) are discussed. Further, the results of two B.Sc (Hons.) theses (Shelton 2003; Mann 2004) that followed this work through the third and fourth year's recovery were summarised. Finally, conclusions on the status of recovery and subsequent recommendations to improve the Present Ecological Status of the aquatic habitats (biotopes) are presented.

7.3.1 Patterns in community composition over two years

Initial analyses made use of multivariate statistical techniques designed to elucidate taxa responsible for the relationships between biotopes and macroinvertebrate community structure from the three sites over the two years. Then the data base of macroinvertebrates was compared to a reference condition developed by Dallas and Day (2007) in order to further investigate whether a satisfactory trajectory of recovery was underway.

Discriminant families for year and biotope groups

The CLUSTER analyses and MDS ordinations (Section 7.2.1) revealed a separation of the aquatic macroinvertebrate assemblages of the mountain stream, at Site 1, from those found lower in the foothill zone, Sites 2 and 3. The ANOSIM test confirmed that the invertebrate communities that occurred at Site 1 were significantly different to those found at Sites 2 and 3, the latter being indistinguishable from one another. The assumption was that zones with the same combination of substrata and flow types would support similar invertebrate assemblages. This finding was corroborated by King and Schael (2001), Schael and King (2005) and Schael (2006), where biotic groupings of macroinvertebrate assemblages were shown to correlated to catchment first, then to channel type (bedrock versus alluvial rivers) and then to position along the longitudinal profile. Here, the correlation was to a combination of channel type and longitudinal position. The aquatic biotopes at Site 1 contained more bedrock than those at Sites 2 and 3 (section 5.2) and are situated in the mountain stream zone, while Sites 2 and 3 occur lower down the longitudinal profile in the foothill zone (Figure 3.3). Overwhelmingly, using the SIMPER analysis, the separation of Site 1 from Site 2 and 3 was attributed to an over-abundance of Amphipoda at Site 1. The Amphipoda comprised 80% and 94% of the total number of individuals collected during the first and second years at Site 1, while only contributing 5% and below to the overall number of individuals collected at Sites 2 and 3. Dallas and Day (2007) reported a frequency of 18% for this taxon in their reference condition data base. The over abundance of Amphipoda at Site 1 is mysterious. It may be partly attributed to the riparian Afromontane Forest community at Site 1, and the subsequent input of coarse particulate matter into the mountain stream. This, combined with the low-flow as a result of the abstraction of water from this section of the river, would create an abundance of palatable vegetative material for these shredders that is not found elsewhere in the catchment. On the other hand, many mountain streams don't have Afromontane Forest (Reinecke *et al.* 2007) so a reference condition data base relevant for Afromontane mountain streams might not be available. Dallas and Day (200) did not report on the status of the riparian vegetation communities of the unidistured rivers from which their reference data base was compiled.

The CLUSTER analyses and MDS ordinations of the site-by-site data appeared to indicate some separation of invertebrate assemblages by the year in which the sample was collected, and also according to the substrata and flow type (section 7.2.1). However this was not confirmed by the ANOSIM tests.

There were no distinguishable differences between the communities found during the first two years. Since samples were only collected once a year it was not possible to re-group them into some other temporal sequence for further analyses. Nonetheless the robustness of the relationship between year-community groups was tested using the ANOSIM pair-wise comparison procedure. None of the habitat descriptor codes, used to identify and describe the habitat available to the macroinvertebrates (Table 4.2), were validated during this analysis. It was not possible to separate out any of the community groups based upon the habitat descriptor codes used to describe the sample: the biotope or combination of flow and substrata type. Similarly it was not possible to separate any of the sample groups based upon the year in which the sample was collected. Thus the habitat descriptor codes were simplified *a posteriori*, as per section 7.2.2.

The habitat descriptor codes (flow and substrata combinations) were reduced to coarser units. Using the ANOSIM procedure, ten community groups were shown to be different. Eight of these were comparisons between the communities found on bedrock and in alluvial substrata from all three sites (Table 7.1). The communities on bedrock were also shown to be different to those from mixed substrata at Sites 1 and 2. The SIMPER procedure was used to determine which macroinvertebrates were responsible for the differences between these pair-wise (habitat) comparisons. In the flow-type comparisons, Chironomidae and Baetidae were shown to be discriminant taxa for the fast-flow types while the Libellulidae were shown to be characteristic of slow-flow types. In the substratum-type comparisons, Amphipoda were shown to be discriminating taxa for the mixed substratum type at Site 1. The Simuliidae, Baetidae and Chironomidae were characteristic of the bedrock community, when compared to those found on mixed and fine substratum types.

Although useful to identify the dominant organisms for each coarse habitat unit (Tables 7.8 and 7.9), SIMPER did not reveal much that could be used to determine whether or not a suitable trajectory of recovery was apparent two years after the fire. Largely this was due to very weak separation of community groups based upon the year of sample collection, the habitat descriptors, or the coarse habitat units. The separation of community groups four years after the fires, based upon the same habitat descriptors, is further discussed in section 7.3.2 along with other relevant conclusions from the subsequent studies of Shelton (2003) and Mann (2004).

Assessing recovery after two post-fire years

Of the 20 species described as representative of Western Cape upland rivers (Dallas and Day 2007), eight were absent after two years (Table 7.10). Three species were always found regardless of site and always in similar proportions. There was an overwhelmingly high presence of Amphipoda at each site but this was most apparent at Site 1. There were slightly higher than expected frequencies of Chironomidae while the Elmidae occurred at the expected frequency. Baetidae were the only other taxon that consistently occurred at higher than expected frequencies. The over abundance of the Amphipoda does deserve further investigation. Otherwise, the pattern was that of lower than expected frequencies for the rest of the reference taxa. Nonetheless, on the basis that 12 of the 20 reference taxa were present in the river after two years, there is good potential for recovery of the macroinvertebrate community. Of the eight missing taxa, seven occur in fast-flowing, clean rivers browsing on algae, periphyton and detritus on rocks, submerged wood or aquatic vegetation (Gerber and Gabriel 2002; de Moor *et al.* 2003; Day *et al.* 2003). The conditions needed for these taxa to establish would thus be an increase in river flow and an increase in the presence of larger substratum particles and submerged woody debris.

Generally there was an increase in pseudo-SASS score and ASPT at the three sites over the two year period (Table 7.11). The pseudo-SASS scores were lowest in the mountain stream at Site 1, higher in the foothill at Site 2 and highest at Site 3. These improvements mean that some recovery had taken place after two years, and this was more evident down the river from Site 1 to 3. The proportion of sensitive versus tolerant taxa was investigated and supported the same conclusion as the pseudo-SASS data. At Site 1, there were far more tolerant organisms present from year 1 to 2 (Table 7.12), indicating a decrease in the quality of habitat available from the first to the second year. At Site 2, the proportion of sensitive and tolerant taxa remained consistent between years. In contrast, there were far more sensitive than tolerant organisms found at Site 3. In terms of invertebrate community structure, there is a trend of decreasing habitat suitability reflected in the decreasing occurrence of sensitive organisms present at Site 1. Moving downstream this is reversed, where no significant changes were shown at Site 2 and increases in the proportions of sensitive taxa reflect an improvement in the habitat suitability at Site 3.

The mountain stream (Site 1) scored more poorly in the Present Ecological Status assessment (Table 7.13) than the foothill (Sites 2 and 3) due to water abstraction from the upper reaches. This reduced flow drastically in the mountain stream until the junction with the first tributary downstream of Site 1 (Reinecke *et al.* 2003). As flow increased downstream, with increasing input from the other tributaries, the impact of water abstraction had less influence on macroinvertebrate community structure.

The mountain stream (Site 1) obtained the lowest pseudo-SASS scores based upon the presence of more tolerant taxa, when compared to Sites 2 and 3. Site 1 also obtained a lower PES score than the other two sites due to abstraction of water that disrupted flow. This could mean that factors other than flow in the lower catchment that are responsible for the higher frequencies of tolerant organisms at Site 2, which are more indicative of disturbance rather than recovery. It is hypothesized that the higher number of tolerant taxa is attributed to the excess of fine sediment being transported downstream each winter, particularly at Site 2 (see Section 5.3). This would retard the ability of flushing flows to sort sediments into their component biotope parts, thus reducing overall physical habitat heterogeneity. This should be investigated further and if it shown to be true, restoration recommendations should include the plans to stabilise the sandy cliffs at Site 2 so that the excess of fine sediments may be flushed downstream with successive winter floods. This would give the aquatic macroinvertebrate fauna of the Silvermine River a better chance to recover toward conditions expected in a reference condition river.

Although the mountain stream (Site 1) invertebrate assemblage was different to that of the foothill Sites (2 and 3), this does not inform on recovery per se. The habitat descriptor codes and their derivatives highlighted some aspects of variation between sites and also habitats between sites. Again, this is not particularly useful for the purposes of assessing recovery. Comparisons against reference data showed that although three taxa were more abundant, 12 of the 20 expected to occur in undisturbed rivers were present in the river after two years. This is positive for further recovery in time. The pseudo-SASS results and PES scores indicated that the mountain stream and the foothill were on different trajectories of recovery. The mountain stream fared more poorly than the two foothill sites, where improved scores reflected an increase in more sensitive organisms after two years.

7.3.2 Patterns in community composition after four years

The data used to assess recovery after four years were collected by Shelton (2003) and Mann (2004) at Site 2 only. These data were formatted for comparison against the reference condition of Dallas and Day (2007). Further, the conclusions these two authors drew based upon the use of habitat descriptors and invertebrate assemblages that are similar to those presented in section 7.3.1, are summarised for the four-year data set (this thesis and the two subsequent honours theses).

The occurrence or absence of a particular taxon, at any given time, may be incidental and take place due to the once-off nature of the sampling methodology, or be as a result of natural fluctuation. Thus, management implications from these findings must be interpreted with caution.

Assessing recovery after four post-fire years

After four years, six of the twenty species expected to occur at Western Cape reference rivers were still absent, an improvement of two species (Table 7.14). Five of these were previously absent after two years, and the sixth new taxon to be absent was Dryopidae. All these taxa should be found in rivers on the Cape Peninsula (Harrison and Agnew 1962, Picker and Samways 1996) but the extent of present day refugia for the recolonisation of these taxa remains unknown and is an avenue for further research. However there remained no increase in the proportion of sensitive taxa, rather there remained an abundance of tolerant organisms. The taxa present during each year changed, with increases in some being countered by losses of others. The three new families that had recruited were Leptophlebidae, Ecnomidae and Blephariceridae. The occurrence of these three families may indicate that the conditions and kinds of biotopes present had improved after four years. The Dryopidae, Helodidae, Limnichidae and Tipulidae, which were present in the first two years, were absent. The proportions of Baetidae and Notonemouridae had reached the expected proportions by the fourth year. The Elmidae had increased beyond the expected frequency after four years. Based upon these results, it appears that no substantial recovery towards a less disturbed state had taken place after four years. The river was still in a disturbed state. Insights into why this may be the case are highlighted from the biotope analyses conducted by Shelton (2003) and Mann (2004) using the full data sets available to them at the time (see below).

The presence of tolerant and sensitive organisms (Table 7.15) remained the same over the four year period. If the river was recovering towards reference conditions there should be a trend of increasing proportions of sensitive taxa over the four years. Since this is not the case it is concluded that the condition of the river remains generally disturbed.

Comparing macroinvertebrate aquatic habitat over four years

During and after the four-year period there remained an abnormally high proportion of fine sediment in the river when compared to other rivers (see Chapter 5). This may have been due to sediment transported into the channel after the fires where the majority of the catchment surfaces were destabilised. On the other hand, this sediment may have been present prior to the fires due to the dominant presence of the woody alien trees on the river banks. Woody alien species are known to cause bank destabilisation that is associated with embedded aquatic habitats (Rowntree 1991). There were no data describing the physical habitat conditions in the river channel prior to the fires or the alien invasion. The activities of the Working for Water personnel may also have contributed towards exacerbating bank instability.

There was some level of correlation between changes in substratum type and the kinds of invertebrates found in the river at that time, yet this is of no particular relevance to recovery. For example, after the high flows during the winters of 2001 and 2002 (Reinecke *et al.* 2003), a large amount of bedrock was exposed in the channel at Site 2. Mann (2004) found a corresponding increase in taxa associated with rocky substrata at this site.

After four years, there was a weak relationship between the invertebrate assemblages in the river and the biotope availability (based upon the habitat descriptor codes). However there remained no justifiable differences between the invertebrate assemblages of the four years, when compared using ANOSIM. The variation about the community groups found in each of the four years was the same. Generally there was a decrease in the proportions of taxa associated with sandy/muddy habitats over the four years. The only anomaly appeared to be a large increase in the proportions of the Corixidae, generally associated with open habitats (Thirion *et al.* 1995), which increased in frequency from 1.5% in 2001 to 71% in 2004. Again this, along with the vast abundance of the Amphipoda, remains to be explained.

There remain very high, largely unstable sandy banks at Site 2 (see section 5.1) which continue to deposit sediment into the channel during high flow events. Until stabilised, this will continue to occur and sediment will continue to be transported downstream. King and Schael (2001) described proportions of aquatic biotopes in four undisturbed foothill Western Cape rivers to have < 2% of wetted sand. By comparison, the proportions of wetted sand over the four-year sampling period at Site 2 made up 47% in 2001 and fluctuated between 20% – 37% thereafter. King and Schael (2001) found that rocky substrata made up 90% of wetted substrata available to macroinvertebrates in undisturbed rivers. By comparison the proportions in the Silvermine River were between 41% and 48%. Mann (2004) found that macroinvertebrates normally associated with stony/rocky habitats, and common to undisturbed rivers (King and Schael 2001), were absent from the Silvermine River. Further, he showed that the common taxa in the Silvermine River at 2004 were those normally associated with sandy/muddy habitats and that these were absent from undisturbed rivers (King and Schael 2001).

The Silvermine River remains disturbed by an excess of sediment in the river channel at Site 2. The high sandy bank present here should be investigated further, as an initial investigation by Akunji (2002) hypothesized the presence of the bank to be a natural feature of a relic floodplain. This could not be confirmed by carbon dating due to budget constraints. Whether or not this is a natural feature, the unstable bank remains a constraint to recovery of the aquatic biotopes and associated macroinvertebrate assemblages. A restoration plan for the river should include recommendations for the grading and stabilisation of this bank. Further it should include a re-vegetation strategy to stabilise the newly graded bank, along with other sections of the river channel (see Chapter 6). Unless this is done, the river in this reach could continue in a degraded state for decades to come. Another constraint to recovery was the diversion of water from the upper reaches, which now has been reduced (Reinecke, *pers. obs.*, January, 2008). The resulting increase in the summer low flow level will benefit all assemblages of macroinvertebrates, particularly those in the upper reaches.

It is clear that without intervention this river will not recover towards levels considered natural and comparable to other undisturbed systems within the foreseeable future. The policy of non-intervention will not successfully restore this system in the short to medium term and it is recommended that the Table Mountain National Park authority consider some measures of active restoration. This should be in line with the principles discussed in Chapter 2 and must involve consultation with a restoration practitioner who has local knowledge of Western Cape rivers.

8 CONCLUSION

River ecosystems that are invaded with exotic plant species are widespread throughout South Africa. There are a number of active participants in efforts to restore these disturbed rivers. Nationwide, restoration activities are being initiated by members of the Working for Water programme housed within the Department of Water Affairs and Forestry, CapeNature, and many researchers in their private capacity as ecological consultants or as researchers at academic institutions. Certainly a restoration mindset currently prevails amongst stewards of South African rivers, within conservation management authorities, academic institutions and in the private sector. However there is no common template of suggested guidelines for river restoration activities that has been specifically tailored for South African rivers. Some initial work was done by King *et al.* (2003), Uys (2003) and Reinecke *et al.* (2007). There is still a pressing need to expose South African river management authorities to international restoration wisdoms. Ultimately, a centralised agency that co-ordinates river restoration activities nationally, collates data and disseminates the lessons learnt country wide is needed.

The success of a restoration project may only be measured against the goals set out before initiation of the project. The goal of the South African National Parks service, the managing authority in charge of the clearing of exotic vegetation from the Silvermine River catchment, was to restore the previously disturbed habitats of the Silvermine River to their undisturbed state. The working hypothesis was that natural processes of recovery (spontaneous succession) were sufficient for recovery to take place. Thus anthropogenic interventions, other than clearing exotic vegetation, were excluded. The question asked in this thesis was, "Can spontaneous succession restore riverine habitats previously disturbed by the presence of exotic plant species?"

This question was posed in three different topics and assessed over the period of one year for each. The state of the physical habitats (biotopes) available to aquatic organisms in the river was measured. Changes in the community composition of riparian vegetation were compared. Similarly, changes to aquatic macroinvertebrate assemblages present in the river were assessed. In each case, conditions in the Silvermine River were additionally compared to reference data bases.

The main findings of each component are summarised next. This is followed by site-by-site management implications. Then, a suggested restoration strategy is outlined for the river.

8.1 General findings

Bank stability

There were no large changes to the macro-channel banks of the river over the one year study. Only one of the foothill cross-sections, cross-section 2-4 at Site 2, showed dramatic change where a meander cut-off occurred. As a result, a large portion of the left bank at Site 2 was eroded. This deposited a large amount of fine sediment into the channel that was transported downstream. A new low-flow channel was forged alongside the un-stable bank, which continues to erode further with each passing winter flood.

Characteristics of flow types

Changes over the year in the proportions of flow at the three sites revealed a lack of faster-flow types, which would normally characterise mountain stream and foothill reaches, at all three study sites. This was attributed to abstraction of water during the summer low-flow period, when the data for this thesis was collected, and the presence of the Silvermine reservoir, which collects summer base flows from the source of the river. Subsequently, the abstraction-weir wall has been broken through and abstraction of summer low-flows halted (Reinecke *pers. obs.* 2007), since Reinecke *et al.* (2003) proposed this change. The reservoir remains in place and since it is a historical feature, being more then 100 years old, and it provides a much needed source of water for combating fires during summer, it is unlikely that it will be removed.

Characteristics of substratum types

Measurements of the proportions of substratum types at the three sites revealed there was an unusually high proportion of smaller substratum types present at each site. Over the year, the three sites reacted differently to the winter floods between the two summer sampling periods.

Riparian vegetation

In the vegetation study, most species that came to dominate the mountain stream (Site 1) and the foothill (Sites 2 and 3) riparian communities emerged from the seed bank within the first two years after the fire. Thus, using the presence or absence of characteristic riparian vegetation species and despite the long recovery time predicted for Riparian Scrub communities (Reinecke *et al.* 2007), it is possible to determine whether there is a need to augment the recovery process after two years. This was shown clearly by the different potential for spontaneous succession to succeed at the three sites.

Aquatic macroinvertebrate assemblages

The aquatic macroinvertebrate communities of the mountain stream differed clearly from those in the foothill. The differences were attributed to a combination of the channel type and the longitudinal position of the sites along the river, both of which dictated the physical habitats available. Further, there was an over-abundance of amphipods in the mountain stream, when compared to the foothills. Investigations into the characteristics of macroinvertebrate assemblages between year and biotope groups did not indicate that recovery had taken place. There were no differences in assemblage groups between the two years nor were the assemblages of different habitats distinct. Although 14 of the 20 expected reference-taxa were present four years after the fires, the general pattern was one of lower abundances than expected. The use of SASS sensitivity ratings to generate pseudo-SASS scores for the three sites in each year showed an increasing improvement in water quality conditions for aquatic macroinvertebrates present from the mountain stream (Site 1) down into the foothill (Sites 2 and 3), and between years. Generally, there were more tolerant taxa found in the mountain stream and more sensitive taxa found in the foothill.

8.2 Constraints to recovery

There were clear differences in the potential for recovery at each of the three study sites. The main constraints to recovery of each site are discussed next.

Site 1

The proportions of finer substrata present at Site 1 after two years had decreased and the corresponding increase in larger particle sizes had approached similar proportions to those found in reference rivers. Further flushing, since the breaking of the abstraction-weir wall, should continue to remove excess fine sediment. The subsequent increase in proportion of larger substratum particles combined with the higher frequency of expected faster-flow types should positively enhance the diversity of the aquatic macroinvertebrates assemblages. A follow up study to map the current state of aquatic biotopes should be done to measure the predicted improvement in habitat quality, due to the increase in summer low-flow.

The riparian vegetation recovered rapidly forming a community dominated by large indigenous trees with an under storey of the expected graminoids and forbs. The structural complexity and dense canopy cover combined with the stability of the banks should provide optimum conditions for the immigration and establishment of other riparian species. On this basis, no active restoration appears to be necessary and it is recommended that the site be monitored every few years to assess changes in community composition and structure. Since recovery of riparian vegetation here fared better than in the foothill, it suggests that areas invaded by *Pinus* spp. may be more resilient than those invaded by *Acacia* spp. This requires further investigation.

Recovery of the mountain stream aquatic macroinvertebrate assemblages was retarded, when compared to the foothill. Since the substratum-type component of the measured biotopes had approached the condition expected of a mountain stream at Site 1, and this was not the case for Sites 2 and 3, it must have been the lack of flow that retarded recovery. Although there was an un-characteristically low level of faster-flow types present at all three sites, this was particularly noticeable at Site1 during the second summer sampling period, where mostly barely flowing and stagnant pools were present throughout the site. Since this study, the abstraction of water from the mountain stream has been halted. There is now water flowing throughout the mountain stream during summer, where previously there were only stagnant pools. Thus, there should be vast improvements in the assemblages of aquatic macroinvertebrates throughout the river, but especially in the mountain stream. A follow up study to investigate the any subsequent changes to aquatic macroinvertebrate assemblages, downstream of the abstraction point, would be interesting now that summer low-flow conditions have been restored.

Site 2

After the winter floods, there remained a high proportion of fine sediment in the channel. This was attributed to the on-going erosion of the un-stable left bank at Site 2 that continues to deposit excessive fine sediment into the channel. This excess of fine sediment is then transported downstream to Site 3 and beyond. As a result, the cobble and boulder riffles and runs, generally associated with Western Cape foothill reaches, are embedded and un-sorted. The status of the physical habitat available to aquatic organisms will remain disturbed unless this bank is stabilised, and then re-vegetated to enhance stability into the future. Further investigations are needed to determine whether this sandy cliff is a relic floodplain feature or whether it exists due to the historical invasion of exotic vegetation. If this is a natural feature, a restoration plan should consider recreating the original constituents of this floodplain and its associated habitats. This may be prohibitively expensive and not consistent with the current demands and pressures on the river ecosystem. On the other hand, if this high sandy bank has accumulated due to the presence of exotic vegetation then it should be dealt with as an exotic itself. A restoration plan that combines bank re-profiling and a re-vegetation strategy would be necessary. If this is done, further deposition of fine sediments will be halted and riparian vegetation may establish naturally on the newly graded bank as vegetative propagules are transported downstream.

The post-fire riparian community consisted mainly of *Acacia saligna* seedlings. As the germinating exotic seedlings were removed via successive follow-up clearing activities, herbaceous species, and to a lesser extent shrubs, successfully recruited. Very few indigenous shrubs and trees germinated, and some of those that did were removed during the clearing of exotics during follow-up clearings. The low structural complexity of the vegetation and infrequent presence of woody taxa may inhibit the potential for future recovery, rendering this site susceptible to future invasion. The presence of woody taxa attracts birds and other mammals, which disperse propagules of other riparian species. Further, mature woody taxa act as sources of seed and vegetative diaspores for the surrounding areas that would otherwise take up to a decade to establish (Reinecke *et al.* 2007). Given the rarity of woody taxa in the post-fire community, the removal of the few individuals that were present during the early stages of recovery could have had a disproportionate impact on the ability of the riparian community to recover on its own. A further complication is the persistence of the exceptionally high cover of the perennial grass, *Ehrharta setacea*. Although this species is present in Western Cape riparian zones it does not normally dominate. The presence of Riparian Scrub and other species in a recovering community may act to suppress the growth of perennial grasses by decreasing incident light levels. This raises the possibility that follow- up clearances, particularly those where indigenous stocks are not protected from the clearing activities, may act to perpetuate the dominance of weedy perennial grasses. The presence or absence of the tree and shrub guilds may be a factor controlling the dominance of weedy grasses in Western Cape riparian zones. Thus re-instating woody taxa could catalyze recovery. Planting indigenous trees may be more successful at catalyzing recovery than sowing seeds in areas where the cover of perennial grasses is high.

Despite the over-abundance of fine sediment and slower than expected flows, there were more sensitive taxa at Site 2, than were found upstream in the mountain stream. This means there are improved conditions of habitat available to aquatic macroinvertebrates, when compared to the mountain stream. This will be further improved if the left bank is stabilised, as this stop continued deposition of fine sediment and allow the subsequent winter floods to flush out the excess. In time, this site could come to resemble a classic foothill that has alternating riffles and runs with their characteristic boulder and cobble beds. This should increase the diversity of taxa favoured by faster-flow over large substrata. If the continually collapsing bank is not stabilised the state of the physical habitat available to aquatic macroinvertebrates will remain in this disturbed state for years, if not decades into the future.

Site 3

After the winter floods, there remained a high proportion of fine sediment in the channel. This was attributed to downstream transport of fine sediment from Site 2. As a result, the cobble and boulder riffles and runs, normally found in foothill reaches of Western Cape rivers, are embedded and un-sorted. The status of the physical habitat available to aquatic organisms remains disturbed.

The vegetation data collected at Site 3 was not analysed for this thesis as it was not possible to separate out the effects of the two different exotic guilds present at this site, viz. *Acacia* spp. (trees) and *Pennisetum clandestinum* (Kikuyu grass). The vegetation of this site was investigated by Reinecke and King (2003). They noted that the state of recovery of indigenous riparian vegetation, here at Site 3, was much the same as that found for Site 2. Thus, the management recommendations proposed to re-establish a cover of indigenous shrubs and trees at Site 2 would apply here. However, the situation as Site 3 is more complicated due to the presence of Kikuyu. The catchment slopes surrounding Site 3 were variously terraced and used for agricultural activities in the past century. Pastures were created so that livestock could graze and subsistence crops and vegetables were grown. Despite this, after the fires there was an established band of non-woody, indigenous Riparian Scrub present. However, this was bordered by a 100% cover of kikuyu from the edge of the macro-channel and outwards up onto the terrestrial catchment slopes. Kikuyu is not an exotic species that is presently being targeted by the clearing activities of the Working for Water programme. In order to re-establish indigenous Riparian Scrub at this site, a buffer zone would have to be established to protect the newly establishing riparian zone from re-invasion by this grass.

Despite the over-abundance of fine sediment and slower than expected flows, there were more sensitive taxa here, than found upstream at Site 2. This means there are improved conditions of habitat available to aquatic macroinvertebrates, when compared to the mountain stream and the upstream foothill at Site 2. Stabilising the left bank at Site 2 should further improve conditions here.

Following these constraints to recovery, a suggested restoration plan for the river is outlined next.

8.3 Restoration recommendations

The common theme revealed by the physical habitat, riparian vegetation and aquatic macroinvertebrate studies was that non-intervention will not support successful recovery in the short to medium term. A blanket policy of non-intervention cannot address site-specific differences, be they natural or artificial, which pose different challenges to restoration. Intervening in the process of recovery in order to facilitate restoration of disturbed ecosystems is in line with modern thinking. Although there is no substitute for the preservation of good quality habitat (Hobbs and Harris 2001), conservation/preservation alone cannot protect riverine ecosystems from the pressures of an expanding human population. Restoration activities are essential in creating sustainable and self-regulating ecosystems. This is not to say that conservation and restoration should be seen as alternative options; restoration activities should work within a framework of sustainable land use and conservation practices (Palmer *et al.* 2006).

It is clear that the mountain stream and the foothill zones of the Silvermine River were recovering at different rates. Thus, they may be said to be on different trajectories of recovery. This means that the time-frame at which natural rates of recovery could proceed, and the likelihood of spontaneous succession to succeed alone, are different in each case. There were zonal differences in the availability of substrata and flow types. There were also other differences that were attributed to water abstraction and the presence of different woody exotic trees. Recovery of the mountain stream is being hampered by the presence of the reservoir and the history of water abstraction (Reinecke *et al.* 2003). Recovery of the foothill is being retarded by erosion from the massive sediment deposit at Site 2. Plans to restore the river should take into account these inter-site differences. The restoration recommendations below are aimed at stimulating aquatic habitat and riparian vegetation recovery. There are no specific recommendations to stimulate invertebrate recovery. Restoring communities of aquatic macroinvertebrates is a little known field and certainly deserves attention. The impact of all of these planned interventions should stimulate further recovery of aquatic macroinvertebrate assemblages in time, as the heterogeneity of biotopes increases with the flushing of the fine sediment, and as the diversity of riparian vegetation improves and matures.

Flow restoration in the mountain stream

- Assess the design and operation of the dam to see if there is a possibility of enhancing flow downstream.
- If possible, consider releasing water from the reservoir during summer to increase flow through the mountain stream, and downstream.
- The increase in flow should stimulate further improvements in the presence of macroinvertebrates characteristic of faster-flow over larger substratum particles.
- Further, increases in summer low-flow should facilitate downstream dispersal of seeds and vegetative propagules from riparian flora that flower during summer and autumn.
- An aquatic ecologist, experienced in the release of environmental flows, should be consulted in this regard.
- Reassess the diversion of water via the abstraction weir with a view to halting all future diversion of water from the river and reservoir.

- Assess the potential for removing the abstraction weir completely. This would allow the channel to recover its natural shape in this section of the river. Reinecke *et al.* (2003) should be consulted in this regard (see Appendix 1).

Bank stabilisation in the foothill at Site 2

- Investigate the origin of the collapsing sandy bank at Site 2. Carbon dating is required, following the recommendation of Akunji (2002), to determine the age and thus whether the sediment plug has accumulated due to the invasion by exotic plants, or whether it is a remnant from a historical floodplain.
- In either case, the unstable bank at Site 2 must be graded to lower its slope. This is necessary so that terrestrial and riparian flora may re-establish.
- An aquatic ecologist with experience in bio-engineering techniques should be contacted in this regard.
- There should be sufficient propagules of indigenous plants supplied from the catchment for natural vegetative recovery to enhance the proposed re-vegetation strategy.
- A re-vegetation strategy would further enhance stability of this bank. A riparian botanist should be contacted in this regard.
- The re-vegetation strategy should initially aim to establish riparian groundcover perennials that will hold the surface layer of sand, and then longer lived riparian shrubs and trees, which have deeper roots to enhance bank stability.
- Reinecke *et al.* (2007) may be used to guide the choice of species and the appropriate location on the river bank that each guild should be established.
- It is recommended that a range of guilds be established i.e. graminoids, herbaceous perennials, shrubs and trees.
- Recommendations for this should follow the guidelines described below.

Riparian vegetation restoration in the foothill

- Continue eradication of exotic plant species, following best practice removal. Holmes (2007) will be particularly useful in this regard. It is vital that indigenous stocks be protected during clearing operations.
- A re-vegetation strategy is also necessary for the riparian community in the foothill. Vegetative recovery may not take place unless woody taxa are re-planted to stimulate the influx of

propagules from upstream and neighbouring catchments. The species list prepared by Reinecke *et al.* 2007 may be used to guide this.

- Since riparian flora occur at characteristic locations on a river bank, in response to elevation above flood levels, it is necessary to establish species from appropriate guilds in the correct location. Failure to do so will result in high mortality of newly established plants, as species specific growth requirements will not be met if plants are established out of their characteristic zones.
    - Graminoids in the Juncaceae and Cyperaceae, are generally found adjacent to the summer low-flow water's edge, within the boundary of the intra-annual flood line. They are adapted to access ground water during the dry summer, and to survive submersion during the wet winter. Species in the genera *Ficinia, Juncus, Carpha* and *Isolepis* (see Appendix 2) would all naturally occur here and may be planted.
    - Restios in the Restionaceae, that may be inter-planted within the intra-annual flood line (along with the graminoids listed above) include *Elegia capensis, Calopsis paniculata* and *Cannamois virgata.*
    - Graminoids in the Poaceae are generally found higher up the bank outside of the intra-annual flood line, where conditions are drier during summer, and the chance of flooding during winter is lessened. Species in the genera *Ehrharta* and *Pentaschistis* (see Appendix 2) would all naturally occur here and may be planted.
    - A mixed community of shrubs and small trees should be planted in mixed clumps at points along the river where their roots may access groundwater during summer low flow. In time, these plants will disperse propagules upstream and downstream.
        - Small trees characteristic of Riparian Scrub are *Metrosideros angustifolia, Brachylaena neriifolia, Brabejum stellatifolium, Morella serrata, Salix mucronata* and *Freylinia lanceolata.*
        - Shrubs characteristic of Riparian Scrub are *Diospyros glabra, Erica caffra, Halleria elliptica, Prionium serratum, Psoralea pinnata* and *Rhus angustifolia.*
- Should the costs of purchasing plants be prohibitively expensive, a seed mix could be prepared and sown in the same locations. However, the success of this would be questionable, as Fynbos plants are notorious for being difficult to germinate. Failure to do so would render the costs spent on seeds wasted. Nonetheless, if this route is followed seeds should be sown during autumn and spring.

Continued monitoring of the state of the physical aquatic habitats, the state of the riparian vegetation communities and the presence of aquatic macroinvertebrates using the methods described in this thesis, will allow tracking of these elements of the riverine ecosystem. If future invasion is prevented, by continually removing germinating exotic seedlings and the influx of exotic propagules, and the restoration recommendations are followed, the mountain stream should continue to recovery further. The foothill will only start to resemble a Western Cape river once the re-vegetation strategy and management of sandy inputs have been implemented. It is unlikely that the riparian vegetation will start to resemble that of undisturbed rivers without the recommended planting and seeding strategy. As these restoration measures are applied, aquatic macroinvertebrates will further inform on the quality of physical habitat.

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10 APPENDICES

Pattern of surface flow in, and abstraction of water from, the Silvermine River

Prepared for the City of Cape Town

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5th December 2003

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1. BACKGROUND TO THIS HYDROLOGY REPORT

Southern Waters Ecological Research and Consulting cc (Southern Waters) was appointed, in association with Erica van den Honert Environmental Consulting cc, by the South Peninsula Administration (City of Cape Town) to formulate an Integrated Catchment Management Plan (ICMP) for the Silvermine River. Ahead of the final compilation of the ICMP, Southern Waters has facilitated the interim preparation of an Action Plan by integrating information collected by different specialists. In addition to the specialist studies, some additional studies were able to further inform the ICMP. One of these was the Silvermine River Rehabilitation Research Project undertaken by Dr Jackie King and Mr Karl Reinecke of the Freshwater Research Unit (FRU), University of Cape Town (UCT). The rehabilitation project was funded by the Water Research Commission (WRC) as part of a larger project jointly awarded to the FRU, UCT and the Department of Earth Sciences of the University of the Western Cape (UWC) (King *et al.* in press).

1.1 Objective and aims of the Silvermine River rehabilitation research project

Objective

The overall objective of the WRC funded Silvermine River Rehabilitation Research Project (SRRRP) was to observe and document the natural process of rehabilitation of the river following devastation of the catchment by fire (January 2000), as well as anthropogenic disturbance associated with clearing of alien vegetation. This work was undertaken during the summers of 2000/2001 and 2001/2002, i.e. spanning one western Cape hydrological year, and recording two successive years of vegetation recovery post the 2000 fires.

Aims

The aims were to record:

- characteristics of the macro- and active channel, and associated aquatic habitat, and to track changes in these and aquatic habitat;
- the nature and re-growth of the riparian and aquatic vegetation;
- details of the assemblages of aquatic invertebrate fauna, thus providing an indication of the health of the river;
- basic water quality variables (pH, EC and T°) as measured with hand-held instruments.

During the second year (2001) of the SRRRP, a meeting of contributors to the Silvermine River Action Plan was convened by Southern Waters. At this meeting (31st May 2001), a decision was made to take advantage of the work already being done along the Silvermine River by contracting the authors of this report to (additionally) measure discharge at monthly intervals along the complete course of the river for one year. The additional work was funded by a grant made available by the South Peninsula Administration (SPA) of the City of Cape Town.

Accordingly the above list of project aims was amended to include:

- an assessment of the present hydrological regime of the Silvermine River, to the extent possible with spot measurements of discharge along its course over at least one hydrological cycle.

1.2 Monitoring discharge in the Silvermine River catchment

The aim of the discharge measurements was to map the main areas contributing to surface runoff and hence flow throughout the catchment. Discharge data were collected at monthly intervals for a period of one year. The hydrological features of the Silvermine River have not previously been recorded. In the absence of a comprehensive hydrological record, the data gathered here were required to inform future discussions between managers and water users on an appropriate flow management policy for the river. The key questions addressed were:

1. What are the main contributing areas of flow in the catchment and how does this pattern change through the year?
2. How much water is diverted from the river at different times of the year?
3. Is abstraction by WLGC and CCC impacting negatively on river flow?

1.3 Further additions to the investigation

After submission of a draft report the City of Cape Town requested additional information on the impact of the diversion weir on the upper end of the system. In particular, input was requested on the status of the length of river between the diversion weir and the confluence with the first tributary. It was further requested that inferences be made about the impact of the dam on the perenniality of the upper Silvermine River, and that the recordings of discharge already made be put into perspective in terms of the WR90 data (Midgley *et al.* 1994). Lastly it was requested that some comment on the present ecological status (PES) of the river be made, drawing on the data from the completed WRC Silvermine River rehabilitation project.

2. RELEVANT CATCHMENT INFORMATION

The Silvermine River is a naturally perennial river approximately 12 km long, with a catchment area of ca. 21 km^2. The Silvermine catchment (Figure 1) is situated within the Winter Rainfall Region, and the river has a mean annual runoff of 4.5x 10^6 m^3 a^{-1} at the reservoir (Silvermine Dam) in the upper catchment (Heydorn and Grindley 1982). Highest rainfalls typically occur from June to August, with February being the driest month. Annual rainfall in the upper catchment averaged 1294 mm between 1908 and 1960, whilst the lower slopes are much drier with Fish Hoek receiving a mean of 663 mm between 1969 and 1981 (Heydorn and Grindley 1982). The winter of 2001 in the south-western Cape Province had more frequent, more intense and more northward tracking cold fronts than many previous winters; July of that year was particularly wet (Reason et al. 2002).

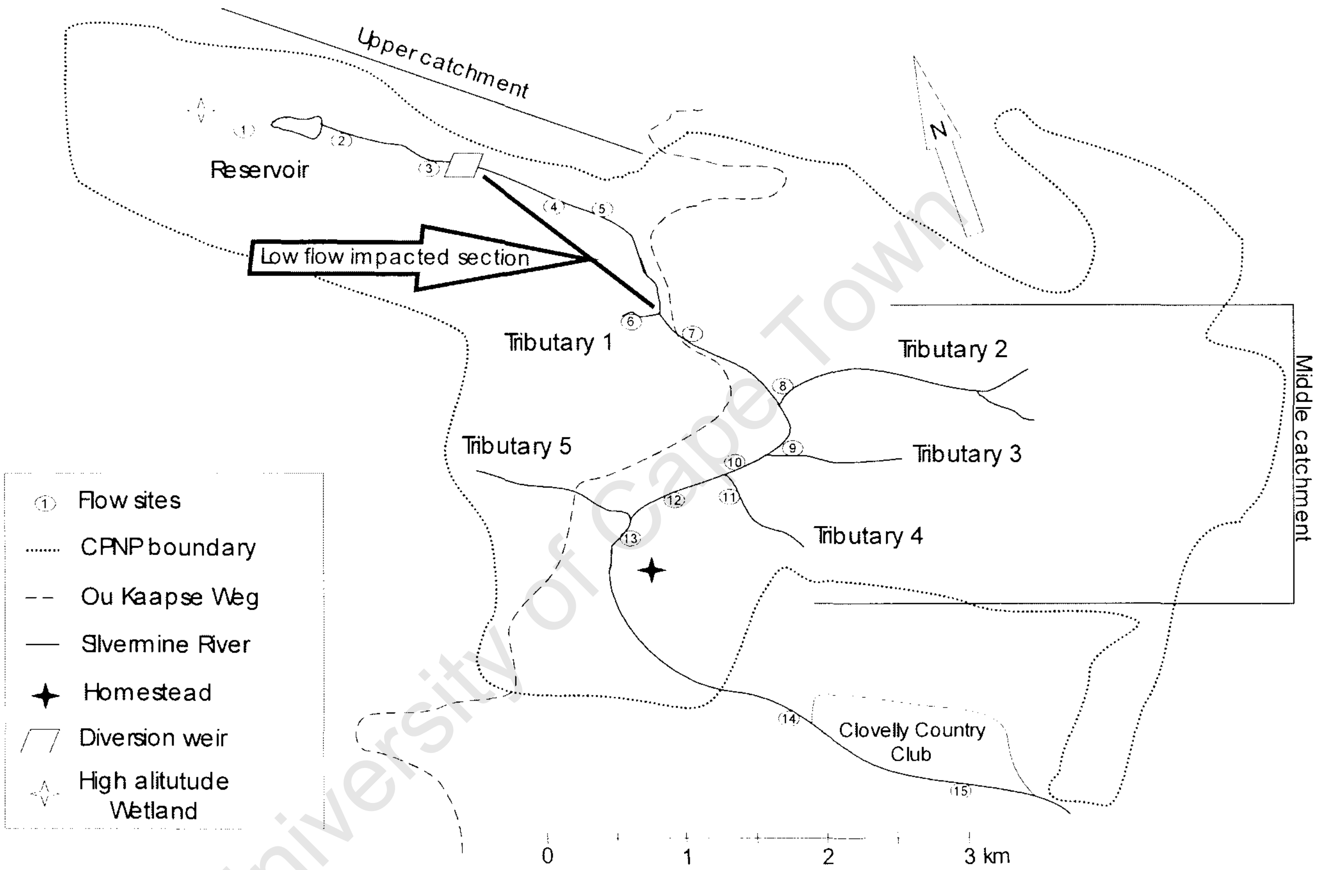

Figure 1 The Silvermine River catchment showing location of the 15 flow sites in relation to present landmarks

The Silvermine Dam is a small impoundment in the upper reaches, just downstream of a high-altitude wetland. The dam was constructed in the late 19th century to provide water for Kalk Bay and Muizenberg. The reservoir has a capacity of 83000 m^3, and fills quickly with a hydraulic retention time of 11 days (Harding 2000). Summer outflows from the dam into the river channel are limited to seepage.

Historically, transfer of water to the then Kalk Bay and Muizenberg municipalities was by means of a diversion weir situated approximately one kilometre downstream from the dam. Currently, and in terms of a long-standing agreement (Appendix 1), this diversion is used to transfer water to the present day Westlake Golf Course. Downstream of the weir, five tributaries appear on maps of the river, three joining the left bank and two the right bank. The channel of the most downstream tributary was not located during this study.

Several houses have been built in the coastal floodplain and a few incorporate the river channel into their gardens. Some residents have boreholes that tap the aquifer of the floodplain. The Clovelly Country Club (CCC), situated adjacent to the floodplain, also uses the river water extensively for irrigation. CCC abstracts water directly from the river by means of a gravity feed that transfers water into a series of three ponds during the winter. These pipes do not appear to remove water during the lower summer flows. There is some indication that the river course may have been changed and restricted in order to expand the fairways and greens. No investigations into the borehole consumption by the residents of Clovelly were undertaken as part of this study.

Urban areas encroaching on the coastal floodplain are prone to flooding during peak rains. A Flood Management Scheme, consisting of a tiered system of wet/dry ponding areas, has been constructed in the lower reaches of the river in order to manage this problem.

2.1 Abstraction of surface runoff by Westlake Golf Course via the diversion weir

A portion of river flow (flow site 3a) is diverted into the weir (Figure 2) where it is joined by a varying volume piped directly from the reservoir. During summer, when flow in the river is low, all river flow is diverted into the weir to join an unknown discharge released from the reservoir. During winter, when there is more river flow, a larger volume is diverted into the weir but some water also flows down the river. Mr Conlin (the greens keeper at WLGC) controls the amount of water released from the reservoir underground to the weir. The total volume of river and reservoir water flowing into the pipe (3b) is available for abstraction at WLGC via the pipe originally used to supply water to Kalk Bay and Muizenburg. Mr Conlin explained that the water flows from the pipe into a balancing tank that maintains the hydrostatic pressure of the sprinkler system at the golf course. During the wet season the tank remains full and water spills out via a high-set outlet into a canal that runs alongside WLGC and empties into Sandvlei. During the dry season, no water spills over into the canal to Sandvlei as the tank is never full. It is not known how much water is used for irrigation as WLGC does not keep records. It was not possible in this study to determine the volumes used by WLGC. It was possible, however, to determine the total amount of water removed from the catchment via the weir (flow site 3b) and then compare this to the amount of water arriving at the tank before WLGC remove any of the water to assess the extent of loss along the way.

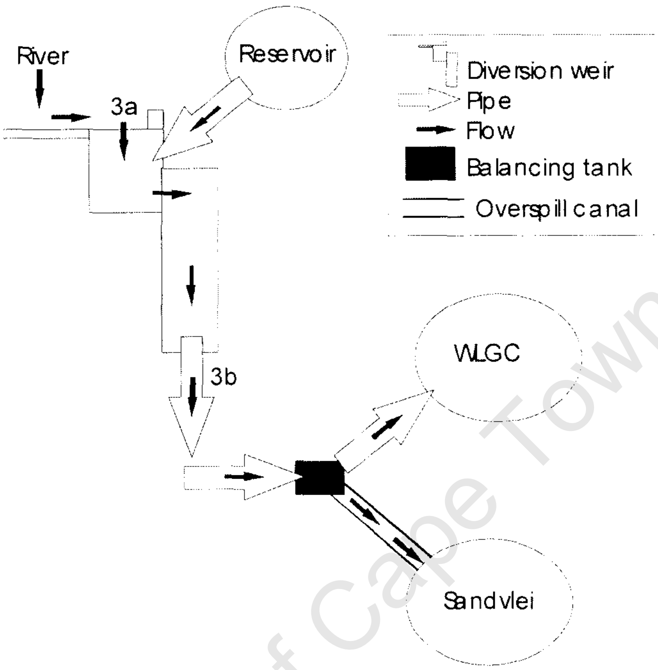

Figure 2 **Schematic of the movement of water through the weir at flow site 3.** 3a = portion of river flow that diverts into the weir. 3b = water diverted into pipe.

3. DISCHARGE MONITORING

3.1 Approach

Fifteen flow sites were chosen in the catchment, at which readings of discharge were taken at approximately monthly intervals from April 2001 to February 2003. The recorded discharges (Appendix 2) are summarised graphically in Appendix 3. Certain flow sites were either dry, or were flowing more slowly than could be measured with our instruments (Table 1), during the low flowing summer period. Flow was recorded at all flow sites during the higher flowing winter months. Six of the flow sites were located in the upper catchment, and the remaining nine in the middle and lower catchments. Four were located on the tributaries. At Flow Site 3, after the meeting with Mr Conlin revealed how flow was being abstracted through the weir, two sets of readings were taken each month; these were the amount of water entering the weir from the river (3a) and the amount of water leaving the weir via the Kalk Bay/Muizenburg pipe (3b). This reading (3b) was compared to the amount of water piped into the tank above WLGC. Monthly discharge readings were taken at all sites on the same day and at least four days after major rainfall events, in order to allow the system to return to fairly stable conditions of flow.

At each site, measurements were taken at each of three cross-sections. Mean (column) velocity and water depth were measured at eight to 15 points across each of the three cross-sections. Velocity was measured using a Marsh McBirney Electronic Flo-Mate current meter on a top-setting wading rod. Aggregate discharge was calculated using the velocity-area method described by Gordon *et al.* (1992). The discharge values calculated for the three cross-sections at each site were averaged to provide a single value per flow site per day. It was not possible to use the electronic current meter to record the rate of flow emptying into the tank above WLGC. This was estimated by taking the average time of five readings to fill a 20-litre bucket.

3.2 Findings

3.2.1 The Silvermine reservoir

The pattern of flow in the Silvermine River was one of high winter flows and low summer flows (Figure 3). Several parts of the drainage system were dry in summer (low flow), or consisted of pools of stagnant water (Table 1). The only tributary consistently recorded to have water all year round was tributary 3. The others were dry during periods of low flow. During these times the channel upstream of the reservoir (Flow Site 1) was also dry, as was a culvert immediately downstream of the reservoir that runs under the road. The first record of water in the main channel of the river occurred some 300m downstream of the dam (flow site 2).

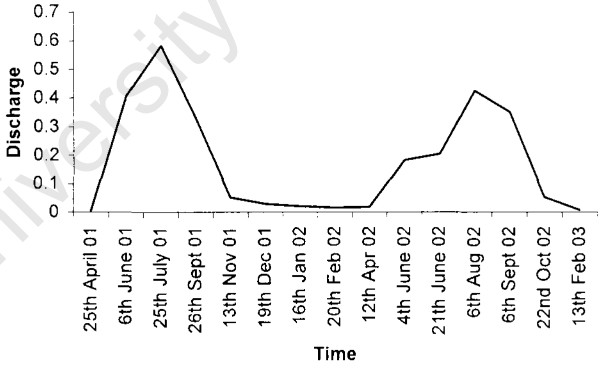

Figure 3 **Hydrological pattern for the Silvermine River as determined from instantaneous measurements of discharge ($m^3 s^{-1}$) at monthly intervals at flow site 15.**

Moving water was observed here on each measured occasion, even though during drier periods the flow was too low to be measured by the flow meter. The presence of water here was presumed to be as a result of seepage from the reservoir. This was not quantified during this study so it is not known what proportion of the total downstream flow results from reservoir seepage. **Since there was water immediately downstream of the reservoir, probably as a result of seepage, and since this volume increased to a measurable flow over the next one kilometre before the diversion weir, it was concluded that groundwater was continually moving down the slopes during the dry season and so this upper section of the Silvermine River would be perennial in the absence of the dam.** A probable scenario in the dry season in the absence of the reservoir, would be a dry channel from the source down to the high-altitude wetland, with underground seepage maintaining the wetland. **Perenniality of surface flow would probably have occurred downstream of the wetland.** The dam has been in existence for over 100 years and the macro-channel downstream, including the riparian vegetation, would by now have adjusted to this. Based on observations of the distribution of peaty soils it seems possible that the high-altitude wetland extended further downstream to the waterfall just downstream of the diversion weir. Based on this assumption it would mean that the dam was built through the middle of what would have been a more extensive wetland, cutting off underground seepage to the areas downstream. This would have led to the destruction of any downstream wetland area. Such a wetland would probably gradually develop again over decades if the dam were removed.

Table 1 **Perenniality of flow sites in summer (during this study).** NPF = no perceptible flow, i.e. current speed <0.001m s^{1}. Flow sites as per Figure 1. Tributaries highlighted.

Flow site	Dry	NPF	Flowing
1	X		
2		X	
3a			X
4	**X**		
5		**X**	
6	X		
7		X	
8	X		
9			X
10			X
11	X		
12			X
13			X
14			X
15			X

3.2.2 Abstraction of surface flow at the diversion weir

The diversion weir is situated 1 km downstream of the reservoir and affects the river differently in different seasons.

Summer low flow periods

Discharge values along the river reveal that the weir had an impact during low flow (Figure 4). **At all times, flow was recorded immediately upstream of the diversion weir but during the drier months no flow bypassed it.** No in-channel flows occurred between the weir (i.e. at flow sites 4 and 5) and the point of confluence with the first tributary (flow site 6; Figure 1). Flow site 4 was dry during periods of low flow. Stagnant pools were found at flow site 5 on each measured occasion. The first tributary, flow site 6, was also dry during these periods of low flow. Flows in the channel downstream of Ou Kaapse Weg (flow site 7) resembled the situation at flow site 2 i.e. the water was observed to be moving but the rate of flow could not be measured with the flow meter. On two occasions in the dry season surface flow had recovered again by flow site 7 (19 December 2002 and 20 February 2002) (Appendix 2 & 3) probably due to seepage from the pools at flow site 5 and from others in the first tributary (flow site 6). This was not the case on the 12th April 2002 and the 13th February 2003, when the river remained without flow down to tributary 2 upstream of flow site 10. On only one occasion was flow recorded downstream of the diversion weir during a low flow summer month (16th January 2002). On this occasion there was more flow in all parts of the river than was previously the case.

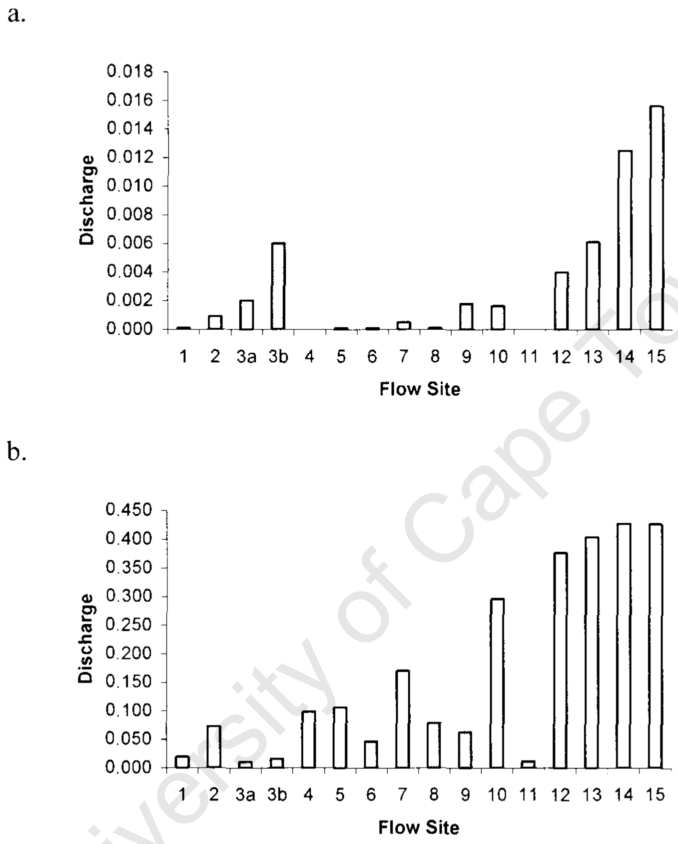

Figure 4 **Discharge at each flow site (m^3 s^{-1}) in (a) a low-flow summer month (20 Feb 2002) and (b) a high-flow winter month (6 Aug 2002).** Tributaries and diversion weir (3b) highlighted.

During 2002 the low-flow impacted section (Figure 1) was dry from February through April (Appendices 2 & 3). During 2003 this same section remained dry from February through July (pers. obs. Dr Jackie King). The most recent observation took place on the weekend of the 6th October (2003) when a small amount of flow was moving over the diversion weir. **In the absence of the weir, the river would be perennial from the reservoir downstream.**

High flow Winter periods

In the winters during this study, when flows were higher water flowed into the weir (flow site 3a) while the remainder over-topped the weir to contribute to the overall downstream increase in discharge (Figure 4). The maximum diverted discharge (flow site 3b) of 0.016 m^3 s^{-1} was recorded on the 6th August 2002, with a maximum discharge of 0.010 m^3 s^{-1} entering the weir from the river (flow site 3a)

(Table 2). **The volume removed from the catchment by WLGC, which is a combination of river and reservoir water (flow site 3b), was approximately the same as that emptying into the tank above WLGC (Table 2, Figure 2) indicating the absence of major leakages in the pipe transporting this water.**

Table 2 **Discharges of water diverted from the Silvermine catchment to WLGC and Sandvlei.** Readings are in $m^3 s^{-1}$.

Reading taken...	6 Aug 02	6 Sep 02	22 Oct 02	13 Feb 03
3a River into weir	0.010	0.006	0.001	0.001
3b Abstraction	0.016	0.011	0.006	0.003
Pipe to balancing tank	0.015	0.011	0.007	0.002
4 Downstream of weir	0.098	0.083	0.008	DRY

3.2.3 Impact of surface flow abstraction by WLGC

The presence of this diversion weir across the channel, and the abstraction of water during low flow periods is a serious impact on the ecological functioning of the Silvermine River. During the WRC Silvermine River rehabilitation project, vegetation and aquatic macro-invertebrate data were collected from the junction of the first tributary and areas downstream of that point. No data, other than the hydrological data, were collected from the areas surrounding the reservoir and the diversion weir specifically to assess their impacts on the river. Thus there are no data on which to base conclusions about expected changes to the riverine flora and fauna of this zone. The following comments on the impacts of these two structures are therefore largely from general knowledge and observation.

The mountain-stream and foothill zones (Flow Site 3 to 7) are most affected by the weir, and the riparian and aquatic parts of the stream ecosystem are probably affected differently. Because there is underground seepage of water, larger riparian plants downstream of both the dam and the weir can maintain themselves through the dry season, although it is highly likely that the extent of riparian vegetation and the number of plant species have declined over the last hundred years. In particular the softer plants, such as sedges, ferns, mosses and annuals, will probably be greatly reduced from their historical levels.

The situation is more critical for the aquatic communities that live in surface waters, however, as they lose all habitat downstream of the weir to the first tributary for much of the year. This could impact the two river zones differently although the results will be much the same. Western Cape mountain streams are typically perennial, with one slow-growing community of small aquatic plants and animals that take most of a year to reach maturity and produce the next generation of seeds, eggs and spores (King 1981). Most species in this community cannot complete life-cycles in the few months of surface water now available downstream of the weir, and so will have disappeared. Any remaining species will tend to be fast-growing, short-lived species that can race through life-cycles when conditions are suitable. The overall result will be a reduction in biodiversity and abundance of native species, with those left commonly seen as 'weedy' or pioneer plant and animal species.

In Western Cape foothill zones, the warmer water and higher nutrient levels support two aquatic communities per year: one in the dry summer months and one in the wet winter months (King 1981). Each community needs the climatic and water conditions of its season and cannot exist in the other half of the year except as eggs, seeds and so on. In the Silvermine River, the riverbed downstream of the weir may dry up too soon for summer species to complete their life cycles and become wet too late in the wet season for winter species to establish themselves and complete their life cycles. Longer-lived species such as dragonflies, damselflies, alderflies, as well as vertebrates such as fish and frogs that need permanent water, cannot survive in this stretch of river at all, unless they move in from areas with surface waters when conditions are suitable. The result, again, must be a substantial decline in biodiversity and biomass of aquatic species compared to historical conditions and, through their loss, less available food for frogs, reptiles, birds and mammals and so fewer of them than might be expected.

The faltering of this stretch of the ecosystem through much of the year will have knock-on affects for the lower reaches of river. Nutrients and individual plants and animals are not drifting downstream, so the lower reaches will be impoverished and less resilient to disturbance than historically. Removal of the weir would return perenniality to the river, re-establish the full suite of aquatic communities and thus greatly enhance biodiversity, food abundance and natural functioning in this system.

3.2.4 The agreement between WLGC and the Cape Town City Council

In light of the new National Water Act, continued abstraction by a current party may only be continued if the user previously had legal rights to abstract. The agreement between the WLGC and the Cape Town City Council (Appendix 1) is a record of the minutes of a meeting held of the waterworks committee on the 30th April, 1957. At this meeting, representation was made by the WLGC for access to water from the Silvermine Reservoir on the basis of the club being in financial difficulty and that the water was not being used for any other purpose by the council. The application was favoured on the basis of some stipulated conditions. These were:

- WLGC to pay an annual fee of £150;
- WLGC to install a connection to the mainline and pay for the installation of the meter (to be supplied by the council);
- WLGC to pay sixpence per thousand gallons, with this and the supply of water being subject to regulations and conditions set by the mayor at the time;
- accounts for the supply to be rendered to WLGC by the council quarterly;
- the maximum amount of water supplied on any one day to not exceed 12000 gallons and to only be drawn during times set by the city engineer;
- water to be delivered only during times of supply through the pipeline and to be discontinued during periods of water shortage;
- the connection to WLGC to remain unconnected with the club's existing spring supply;
- the agreement to be subject to termination with 12 months notice on either side.

The mayor, town clerk and a representative from WLGC were all witness to the arrangement. It is not known whether WLGC has continued to meet the terms as set out in the minutes or whether or not this document allows WLGC legal access to the water in perpetuity. Mr Les Conlin, the greenskeeper at WLGC, did inform Mr Reinecke that there are no records of the volumes of water being used by WLGC. Since the volumes abstracted at the diversion weir are approximately the same as those arriving at WLGC (Table 2), there does not appear to be leakage of abstracted water as it is transported to WLGC.

Part 2 of the National Water Act (NWA 1998) "...requires every catchment management agency to progressively develop a catchment management strategy for the water resources within its water management area". This will require the catchment management agency to "...seek co-operation and agreement on water related matters from the various stakeholders and invested persons" (NWA 1998), where all currents users and any impediments to the water resource must be assessed. Continued use of a water resource may continue under a valid license, until such time as a responsible authority requires such a user claiming entitlement for the purposes of abstraction, to apply for a licence (NWA 1998). Since the majority of the mountain-stream zone of this river lacks flow during dry periods, as a result of abstraction of this water to WLGC, and the fact that this river is in a National Park, a case could be made for reinstatement of flows along the whole river.

3.2.5 Abstraction from the river by the Clovelly Country Club

An indication of the level of abstraction by CCC is given by comparing the readings at flow sites 14 (upstream of CCC) and 15 (downstream of CCC) (Figure 1, Table 3). The volumes of water downstream of CCC were either larger, or the same, than those recorded upstream of CCC (Table 4). In the four instances where a larger volume was recorded upstream (Flow Site 14), the value fell within (and in one case slightly above) the 95% confidence limits defined for the reading downstream (Flow Site 15). In summer, no water was observed flowing into their abstraction pipe (which is sited above the low-flow water level) and during winter there is sufficient water in the river to fill their ponds without their removing a large proportion.

Table 3 **Average (± SD) of three measurements of discharge upstream (Flow Site 14) and downstream (Flow Site 15) of CCC over one annual hydrological cycle.** Discharge values ($m^3 s^{-1}$) recorded upstream CCC that were higher than the comparative reading taken downstream are italicised.

Reading taken	25 Jul 01	26 Sep 01	13 Nov 01	19 Dec 01	16 Jan 02	20 Feb 02	12 Apr 02	4 Jun 02
Flow Site 14	0.533	*0.340*	*0.067*	0.024	*0.024*	0.012	0.012	*0.203*
STDDEV	0.013	0.018	0.004	0.003	0.004	0.002	0.001	0.015
Upper 95% CI	0.546	0.359	0.071	0.027	0.028	0.015	0.013	0.218
Lower 95% CI	0.521	0.322	0.064	0.021	0.019	0.010	0.011	0.188
Flow Site 15	0.583	0.327	0.060	0.029	0.021	0.016	0.017	0.182
STDDEV	0.131	0.015	0.016	0.004	0.004	0.004	0.002	0.020
Upper 95% CI	0.714	0.342	0.076	0.033	0.026	0.020	0.019	0.202
Lower 95% CI	0.452	0.312	0.044	0.024	0.017	0.011	0.015	0.162

Abstraction by the CCC does not have a detectable impact on surface flow, even during low flowing periods. During the winter months discharge in the river at CCC is large enough to render the amount abstracted negligible.

4. SCALING DOWN THE WR90 DATA FOR THE SILVERMINE CATCHMENT

4.1 Approach

In the absence of a gauged hydrological record for the Silvermine catchment, a combination of the recorded discharges and scaling down of the WR90 data (Midgley *et al.* 1994) by area provides rough estimates of average monthly flow. Midgley *et al.* (1994) provided a simulated natural river flow series for tertiary catchment G22. The quartenary catchment G22A, which encompasses the Silvermine River and the rest of the Cape Peninsula south of Hout Bay, is situated within the boundaries of tertiary catchment G22. A single MAR value is given for G22A. The average monthly flow estimates for G22, as presented in Appendix 7.2 of the WR90 report, were apportioned to the Silvermine catchment by calculating the relative percentage each monthly flow volume contributes to the G22 MAR, than transposing these monthly percentages to the G22A MAR (for the peninsula). The G22A monthly averages were than scaled down by area to given monthly MAR values for the Silvermine catchment.

4.2 Findings

The simulated values represent the *average* runoff expected over a year for a river on the Cape Peninsula (Table 4). The recorded flows, as calculated from the spot measurements of discharge show the same trends over the year as the simulated values, namely higher flows in winter and lower flows in summer. **The MAR recorded during this study is much greater than the simulated MAR.** This is largely due to higher runoff during the recorded wet season. Since readings of discharge were only taken at least three to four days after rainfall events, it is likely that the *actual* MAR during this study period was much higher than those *recorded* since flood discharges were purposely missed. **Therefore both the simulated flow series estimated from the WR90 data (Midgley *et al.* 1994) and the monthly average totals recorded during this study underestimate the *actual* MAR during the study period.** The simulated flow series presented by Midgley *et al.* (1994) is thus not suitable for a flow management programme. The production of both sets of values, as presented in Table 4, has inherent inaccuracies. The WR90 report gives natural simulated riverflow for tertiary catchment G22 only, an area totalling 1622 km^2, which encompassed the whole peninsula and the catchments of the Kuils, the Eerste, the Lourens and the Sir Lowry's Rivers. The Silvermine catchment may or may not share many similar characteristics with the averaged out G22 parameters used in running the WR90 simulation. This limits the applicability of the G22 simulated runoff values to the Silvermine catchment. Similarly the use of spot measurements of discharge on an annual cycle to provide monthly average discharge may also grossly misjudge MAR since significant discharges would have been missed as readings were only taken three to four days after rainfall events.

Table 4. **Recorded (this study) and simulated (WR90) monthly average discharge (m^3 s^{-1}) and runoff (10^3 m^3) for the Silvermine River catchment.** Simulated data have been disaggregated by area from the WR90 data for the Cape Peninsula, in particular from the simulated natural river flow given in Appendix 7.2 (Midgely *et al.* 1994). The recorded values used are those recorded at Flow site 14 (30 m upstream of the bridge crossing the river to the par 3 green), upstream of where CCC abstracts water from the river.

This study	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Ave	MAR
Q	0.052	0.067	0.024	0.024	0.009	0.009	0.012	0.150	0.290	0.533	0.427	0.303	0.158	
Runoff	139.3	173.7	64.28	64.28	21.77	24.11	31.10	401.8	751.7	1427.6	1143.7	785.4		5029
WR90														
Q	0.085	0.046	0.019	0.008	0.007	0.006	0.028	0.080	0.160	0.213	0.242	0.159	0.088	
Runoff	226.5	120.1	51.04	21.91	16.72	16.72	72.60	215.5	414.8	570.9	649.1	412.1		2788

5 PRESENT ECOLOGICAL STATUS (PES)

5.1 Approach

An assessment of PES takes into account the number and severity of anthropogenic disturbances in river catchments. A river is classed according to an overall score indicating the potential for degradation caused by perturbations in the catchment. These disturbances "include abiotic factors, such as water abstraction, weirs, dams, pollution and dumping of rubble, and biotic factors, such as the presence of alien plants and animals" (Dallas 2000). Each assessed category is given a score and each category contributes a weighted percentage toward a total score. PES scores for the Silvermine River were calculated in line with the procedures required for assessing Habitat Integrity Status (Dallas 2000). The method generates in-stream and riparian status scores separately. The Silvermine catchment was divided into five representative reaches, each of which was assessed for PES according to the guidelines prepared by Brown *et al.* (2001). These guidelines were designed to assess PES in the absence of hydrological records for rivers in the Western Cape. The representative reaches were:

1. Reach 1 (source to diversion weir; between Flow Sites 1 and 3);
2. Reach 2 (diversion weir to first tributary; between Flow Sites 3 and 6);
3. Reach 3 (first to second tributary; between Flow Sites 6 and 8);
4. Reach 4 (second tributary to Sunbird weir; between Flow Sites 8 and 13); and
5. Reach 5 (Sunbird weir to mouth; between Flow Sites 13 and 15).

The calculated scores were used to classify the assessed river reaches as A to F class PES according to the percentages obtained for each reach (Table 5).

5.2 Findings

Reaches 1 and 2 scored poorly (class C) (Table 6). This was due to the presence of the reservoir and the abstraction of water by WLGC. Reach 3, presently classed as B, was impacted by the lack of flow during the dry season as a result of the abstraction upstream. Further downstream in the valley, downstream of Flow Site 8, the presence of alien vegetation, notably *Acacia saligna* (the Port Jackson), *Populus canescens* (the grey poplar) and *Pennisetum clandestinum* (kikuyu grass) impacted negatively on the riparian score and status. In these two lower reaches (Flow sites 8 to 15) the impact of the diversion and the reservoir are lessened. As a result the In-stream status and score is much improved (class B), while the riparian score drops lower due to the alien vegetation.

Accepting that the dam will remain, the PES scores of reaches 1 to 3 would best be improved by halting diversion of water from the river into the weir during the dry season and even, maybe, by releasing some water from the dam for downstream river maintenance. The continual clearing of alien vegetation and subsequent re-growth of indigenous vegetation will improve the PES of reaches 4 and 5.

Table 5 **Preliminary present status classes** (Kleynhans 1996, cited by Dallas 2000).

Class	Description Score	(% Of Total)
A	Unmodified, natural.	
B	Largely natural with few modifications. A small change in natural habitats and biota may have taken place, but the assumption is that ecosystem functioning is essentially unchanged.	80 - 89
C	Moderately modified. A loss of change in natural habitat and biota has occurred, but basic ecosystem functioning appears predominately unchanged.	60 - 79
D	Largely modified. A loss of natural habitat and biota and a reduction in basic ecosystem functioning is assumed to have occurred.	40 - 59
E	Seriously modified. The loss of natural habitat, biota and ecosystem functioning is extensive.	20 - 39
F	Modifications have reached a critical level and there has been an almost complete loss of natural habitat and biota. In the worst cases, the basic ecosystem functioning has been destroyed.	0 - 19

Table 6. **Present ecological status scores and classes for five representative river reaches.** The five representative reaches are located according to Flow site (Section 5.1, Figure 1).

Flow sites	Reach 1	Reach 2	Reach 3	Reach 4	Reach 5
In-stream status score	66.6	67.3	83.6	82.6	83.2
Riparian status score	70.7	73.8	80.5	70.3	57.6
In-stream status class	C	C	B	B	B
Riparian status class	C	C	B	C	D

6. SUMMARY OF THE IMPORTANT FINDINGS

Appendix 3 summarises all the discharge readings taken at the 15 flow sites during this study. The measured flows were grouped into seven categories and plotted on successive maps of the main channel and the measured tributaries. From the maps the following is clear:

- The general pattern is that flow occurs in all tributaries and all parts of the main channel during winter. During summer the tributaries were dry, stagnant or barely flowing.
- No flow was recorded in the main channel upstream of the reservoir during the dry season. No flow was observed to be moving through the culvert under the road downstream of the dam during low flow periods.
- There was moving water 200m downstream of the reservoir, even though the flow meter could not quantify the amount. In part this is due to seepage from the reservoir during the dry season. This section would probably be perennial in the absence of the dam.

- Flow was recorded in the main channel immediately upstream of the weir on all occasions, irrespective of season. This means that flows would have occur downstream of this point all year in the absence of the weir, during the study period.
- During summer all this flow was abstracted to WLGC via the pipe from the weir. This impacted flow at least to flow site 7 and on two occasions to flow site 10.
- During times of low flow, a larger volume of water was abstracted through the weir than was recorded upstream of the weir, the extra water being piped down from the reservoir.
- The reading upstream (flow site 14) and downstream of CCC (flow site 15) are the same (Table 3) indicating the golf course had no major impact on the flow volume.
- The MAR recorded during this study is much greater than the simulated (WR90) MAR. The simulated flow series estimated from the WR90 data (Midgley *et al.* 1994) and the monthly average totals recorded during this study underestimate the *actual* MAR during the study period. The simulated flow series (WR90) is thus not suitable for a flow management programme.
- The upper catchment (Flow sites 1-6) has a PES of C, that is, it is moderately modified. This is due to the abstraction and the presence of the reservoir. The next reach downstream (Flow sites 6 to 8) is in the best condition: class B (i.e. largely natural). The rest of the river downstream through the valley and past the Sunbird weir is somewhat impacted by the dam and weir, but its low riparian score of C and D is mainly due to alien vegetation.

7. CONCLUSION

The upper Silvermine River was in all likelihood perennial prior to the construction of the dam in the late 19th century. Abstraction of water at the diversion weir is the single most damaging impact affecting the ecological functioning of the whole river. The entire mountain stream zone ceased to flow during summer or other low flowing periods during this study because all available flows were abstracted to WLGC at the diversion weir. Flow only recovered in the main channel after inputs from ground water or a flowing tributary. Abstraction of surface flow by CCC does not impact negatively on river flow. Water is only abstracted during the winter, or other high flowing periods.

8. RECOMMENDATIONS

Abstraction of water at the weir should be reviewed. If water is still to be abstracted, it is recommended that no water be abstracted during summer or other periods of low flow. Three possible options in increasing order of complexity and expense (due to construction, manpower or otherwise), are:

1. halt all abstraction of river water throughout the year;
2. allow abstraction of water during the wet season or other periods of high flow;
3. remove the weir.

In order to stop all abstraction, the diversion weir wall situated in the main channel, which deflects water from the channel into the filter baskets, would need to be removed. This is the simplest of the three options by far since it requires a minor alteration to the existing structure, to allow all flow in the channel at site 3 to proceed down the river at all times of the year. This could probably be done without tampering with the entire concrete structure. Water could still be transferred from the reservoir down through the diversion structure and to WLGC if so required. There would be a minor disturbance associated with the removal of the weir wall. This may require stabilisation of the river bed to prevent development of a nick point that would erode upstream.

If abstraction of river water is to be continued during periods of high flow the weir needs to be modified to allow low flows past the weir during the dry season. At present all low flows are diverted into the pipe down to WLGC. This will require further construction that will add to and further disturb the river but will provide benefit in terms of summer flows remaining in the river. Management of abstraction in the wet season will require control and manpower to facilitate the opening and closing of the abstraction weir.

Removal of the weir and all its associated structures is the most expensive and complex option. To remove the entire structure would probably require an EIA. A fluvial geomorphologist, riparian botanist and at least one structural or hydraulic engineer would be essential for such an exercise. Once removed, it would be essential to stabilise and re-vegetate the newly designed channel in the cleared area.

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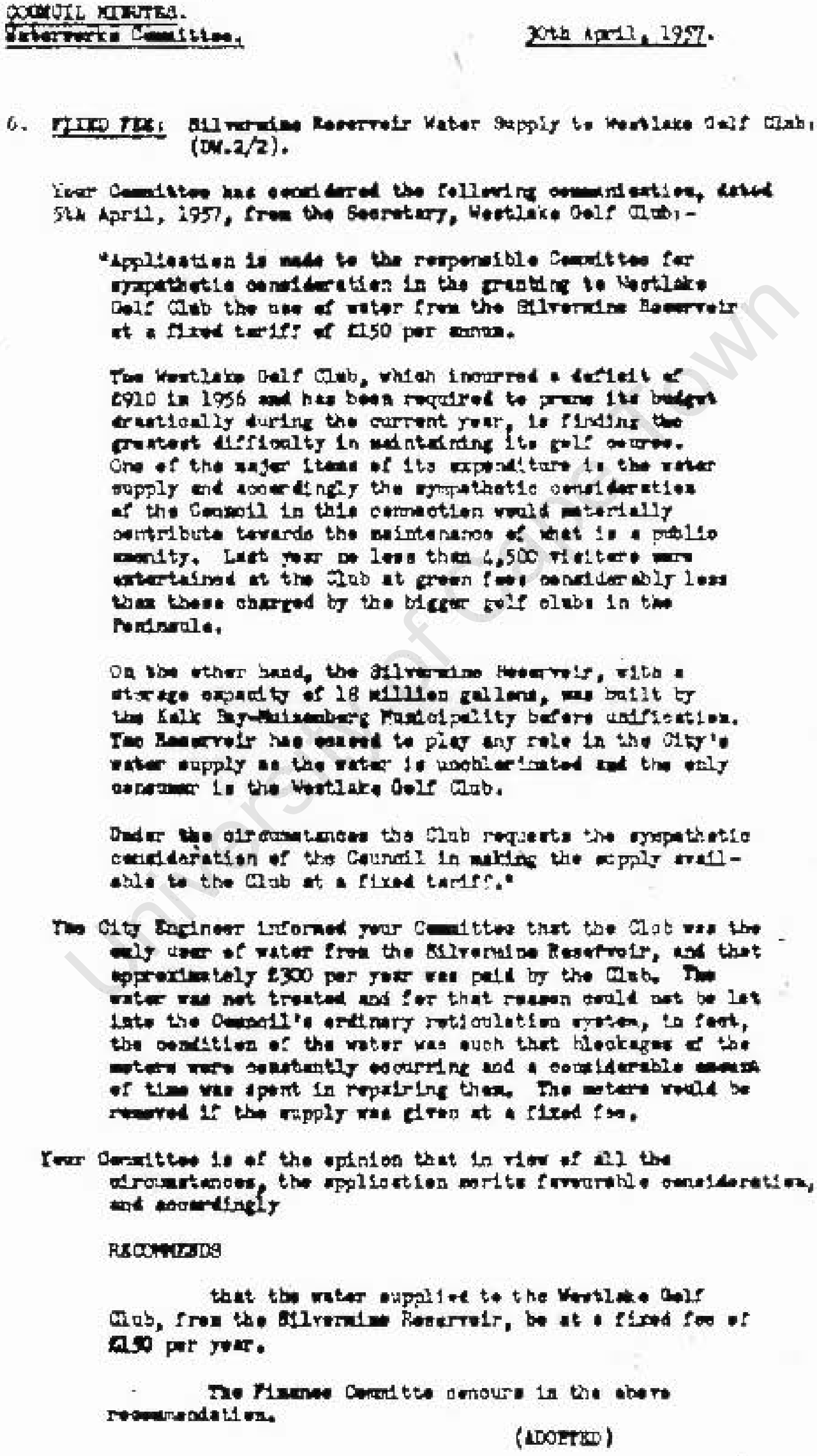
Appendix A A copy of the minutes outlining the terms of the agreement between Westlake Golf course and the Cape Town City Council.



COUNCIL MINUTES.
Waterworks Committee. **30th April, 1957.**

6. **FIXED FEE:** Silvermine Reservoir Water Supply to Westlake Golf Club; (DW.2/2).

Your Committee has considered the following communication, dated 5th April, 1957, from the Secretary, Westlake Golf Club:-

> "Application is made to the responsible Committee for sympathetic consideration in the granting to Westlake Golf Club the use of water from the Silvermine Reservoir at a fixed tariff of £150 per annum.
>
> The Westlake Golf Club, which incurred a deficit of £910 in 1956 and has been required to prune its budget drastically during the current year, is finding the greatest difficulty in maintaining its golf course. One of the major items of its expenditure is the water supply and accordingly the sympathetic consideration of the Council in this connection would materially contribute towards the maintenance of what is a public amenity. Last year no less than 4,500 visitors were entertained at the Club at green fees considerably less than those charged by the bigger golf clubs in the Peninsula.
>
> On the other hand, the Silvermine Reservoir, with a storage capacity of 18 million gallons, was built by the Kalk Bay-Muizenberg Municipality before unification. The Reservoir has ceased to play any role in the City's water supply as the water is unchlorinated and the only consumer is the Westlake Golf Club.
>
> Under the circumstances the Club requests the sympathetic consideration of the Council in making the supply available to the Club at a fixed tariff."

The City Engineer informed your Committee that the Club was the only user of water from the Silvermine Reservoir, and that approximately £300 per year was paid by the Club. The water was not treated and for that reason could not be let into the Council's ordinary reticulation system, in fact, the condition of the water was such that blockages of the meters were constantly occurring and a considerable amount of time was spent in repairing them. The meters would be removed if the supply was given at a fixed fee.

Your Committee is of the opinion that in view of all the circumstances, the application merits favourable consideration, and accordingly

RECOMMENDS

that the water supplied to the Westlake Golf Club, from the Silvermine Reservoir, be at a fixed fee of £150 per year.

The Finance Committe concurs in the above recommendation.

(ADOPTED)

AGREEMENT made and entered into this 7th day of December One Thousand Nine Hundred and Forty-nine between the **COUNCIL OF THE CITY OF CAPE TOWN** (hereinafter referred as "the Council") of the one part, and THE WESTLAKE GOLF CLUB (hereinafter referred to as "the Club". of the other part.

WHEREAS the Club has applied to the Council for a supply of water and, by virtue of its rights as riparian owner of the Silvermine Catchment Area, has requested some concession to be granted to it by the Council relative to the supply of such water.

AND WHEREAS the Council has agreed to supply water as desired by the Club subject to the terms and conditions herein set forth.

NOW THEREFORE THESE PRESENTS WITNESS:-

1. The Club shall pay the cost of connection to the Council's main at the nearest point to the Club's boundary and the installation of a branch leading with the necessary stop valves and the installation of the meter which will be supplied by the Council free of charge.

2. The Club shall pay for water supplied in accordance with the registration of such meter at a charge of sixpence (6d.) per thousand gallons of water consumed and the supply of water and payments therefor shall be subject to the regulations and conditions (so far as the same may be applicable) in force in the City of Cape Town from time to time.

3. Accounts for water supplied under this agreement shall be rendered quarterly by the Council. All notices and accounts shall be addressed to the Westlake Golf Club at its address, P.O. Box 2, Muizenberg, and accounts shall be payable at the office of the City Treasurer, Electricity House, Strand Street, Cape Town, free of exchange.

4. The maximum quantity of water to be supplied by the Council under this agreement, during any one day, shall not exceed twelve thousand (12,000) gallons unless otherwise authorised by the City Engineer and shall be drawn only during such periods as shall be laid down from time to time by the City Engineer.

5. Water shall be delivered only during such time as a supply is passing through the Council's pipe line and the supply shall be discontinued by the Council during periods of water shortage in the municipal Area, notification of which will be made to the Club by the City Engineer.

The connection now being provided from the 12" main shall be utilized solely to supply the lower part of the Golf Course and there shall be no interconnection between the Council's supply and the Club's existing spring supply. No stoppage, reduction or decrease in the pressure or supply of water shall in any way render the Council liable for any damage which may accrue by reason thereof

6. This agreement may be terminated on 12 (twelve) months' notice given by either party at any time.

7. This agreement shall become effective only after the Third Steenbras Pipe line has been brought into use.

The Common Seal of the City
of Cape Town was hereunto
affixed at the City Hall,
Cape Town, this 7th day
of [illegible], 1949 in the
presence of

(sgd) Thos. O. B[illegible]

MAYOR.

Certified as being in terms
of the resolution of the
Council dated 30th June, 1949.

(sgd) M. A. Williams

TOWN CLERK.

Signed for and on behalf of
the Westlake Golf Club by
(sgd) V. E. D[illegible]

AS WITNESSES:

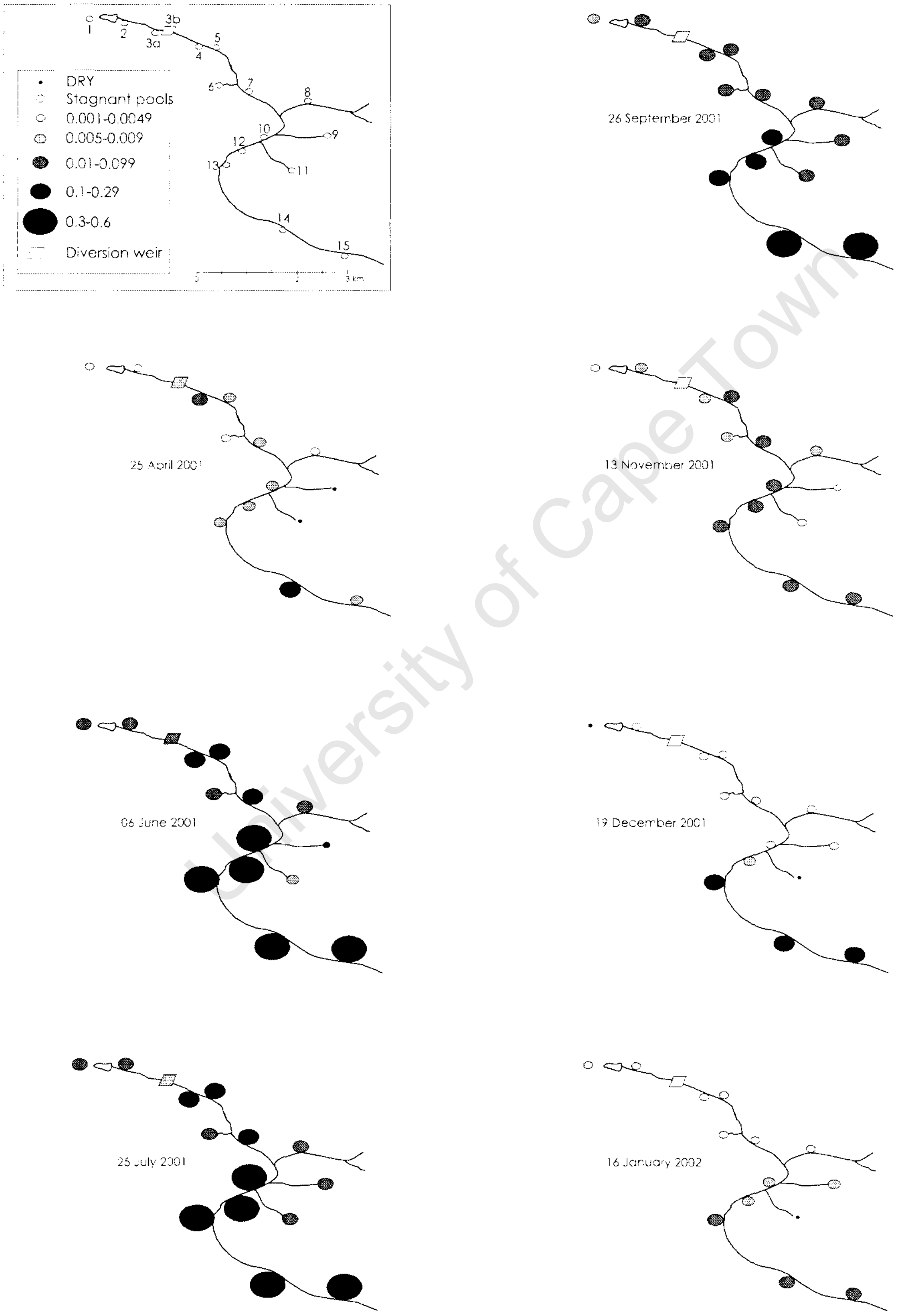
1. (sgd) Ron Walker ....

2. (sgd) E. Wall ....

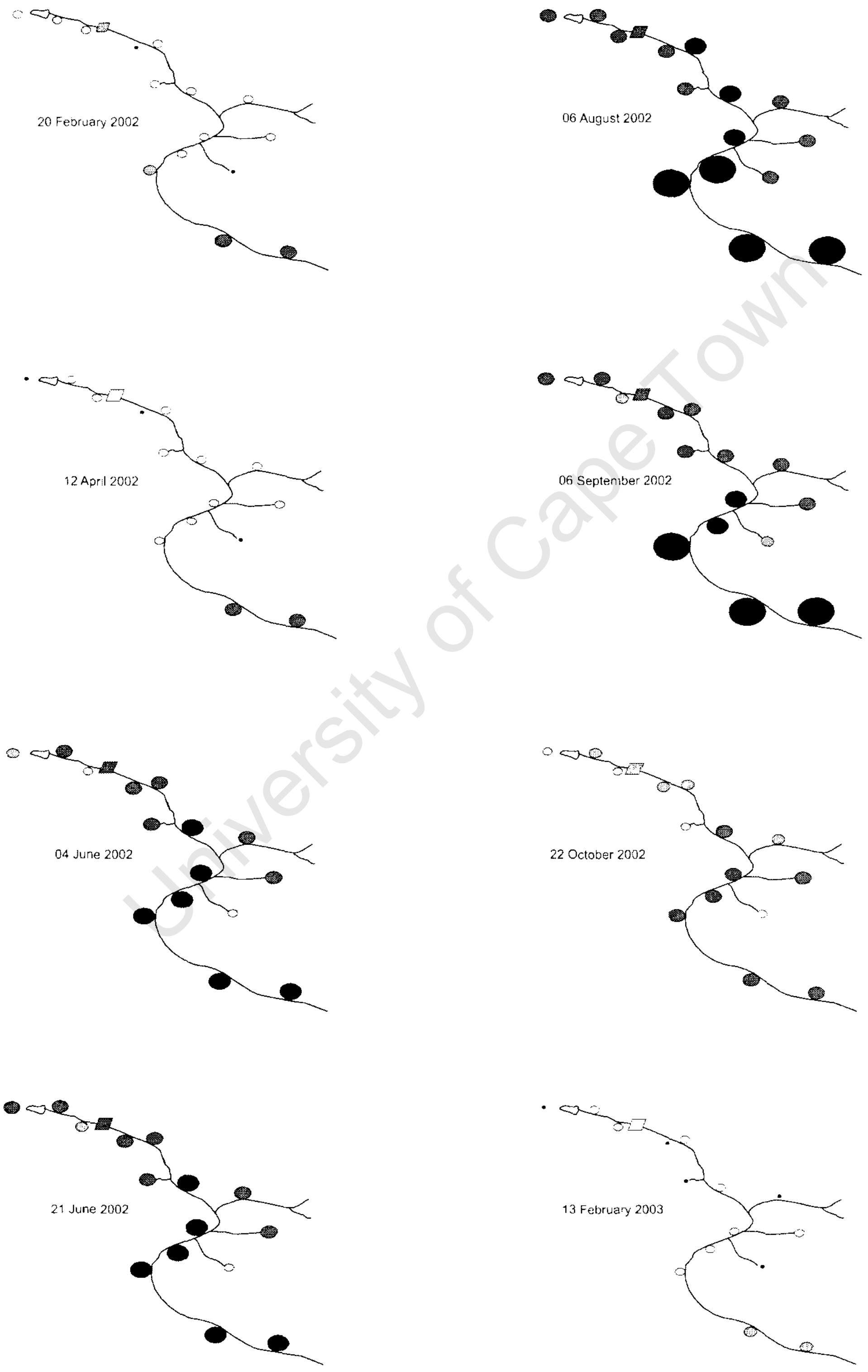
Appendix B Discharge readings taken at the 15 flow sites. Values are in $m^3 s^{-1}$. NPF = no perceptible flow, i.e. current speed <$0.001 m s^{-1}$. At flow site 3 the volume of water entering the weir from the river (3a) is not the same as that diverted along the pipeline (3b) since the water from the river is joined by a volume piped from the reservoir (the difference between 3a and 3b). Tributaries and diversion highlighted. # = no reading taken.

Flow site	25 Apr 01	6 Jun 01	25 Jul 01	26 Sep 01	13 Nov 01	19 Dec 01	16 Jan 02	20 Feb 02	12 Apr 02	4 Jun 02	21 Jun 02	6 Aug 02	6 Sep 02	22 Oct 02	13 Feb 03
1	0.002	0.033	0.017	0.006	0.002	**DRY**	0.002	**NPF**	**DRY**	0.007	0.010	0.019	0.014	0.001	**DRY**
2	**NPF**	0.095	0.071	0.041	0.007	**NPF**	0.001	0.001	**NPF**	0.052	0.037	0.072	0.055	0.006	**NPF**
3a	#	#	#	#	#	#	#	0.002	0.002	0.004	0.009	0.010	0.006	0.001	0.001
3b	**#**	**#**	**#**	**#**	**#**	**#**	**#**	**0.006**	**0.004**	**0.012**	**0.014**	**0.016**	**0.011**	**0.006**	**0.004**
4	0.011	0.126	0.148	0.045	0.006	**NPF**	0.001	**DRY**	**DRY**	0.074	0.068	0.098	0.083	0.008	**DRY**
5	0.005	0.158	0.124	0.060	0.010	**NPF**	0.001	**NPF**	**NPF**	0.072	0.069	0.106	0.087	0.008	**NPF**
6	**NPF**	**0.056**	**0.063**	**0.032**	**0.007**	**0.001**	**0.001**	**NPF**	**NPF**	**0.020**	**0.022**	**0.046**	**0.034**	**0.002**	**DRY**
7	0.005	0.204	0.223	0.088	0.015	0.002	0.003	0.001	**NPF**	0.138	0.112	0.170	0.141	0.014	**NPF**
8	**NPF**	**0.077**	**0.082**	**0.045**	**0.007**	**NPF**	**0.001**	**NPF**	**NPF**	**0.027**	**0.026**	**0.078**	**0.052**	**0.008**	**DRY**
9	**DRY**	**NPF**	**0.039**	**0.018**	**NPF**	**0.003**	**0.008**	**0.002**	**0.001**	**0.027**	**0.025**	**0.063**	**0.044**	**0.023**	**0.004**
10	0.008	0.440	0.446	0.221	0.027	0.004	0.007	0.002	0.001	0.177	0.138	0.296	0.217	0.034	0.002
11	**DRY**	0.006	0.018	0.011	0.001	**DRY**	**DRY**	**DRY**	**DRY**	0.001	0.001	0.011	0.006	**NPF**	**DRY**
12	0.008	0.452	0.573	0.272	0.043	0.007	0.007	0.004	0.002	0.223	0.135	0.377	0.256	0.034	0.001
13	0.007	0.410	0.605	0.294	0.043	0.014	0.013	0.006	0.003	0.190	0.236	0.405	0.320	0.054	0.003
14	0.012	0.502	0.533	0.340	0.067	0.024	0.024	0.012	0.012	0.203	0.166	0.427	0.297	0.052	0.006
15	0.005	0.406	0.583	0.327	0.060	0.029	0.021	0.016	0.017	0.182	0.204	0.426	0.315	0.052	0.007

Appendix C **Catchment maps showing the measured flow at 15 sites over 15 months.**
Key block: numbers = flow sites, discharge is in $m^3 s^{-1}$



Appendix C (contd.) **Catchment maps showing the measured flow at 15 sites over 15 months.** Key block: numbers = flow sites, discharge is in m^3 s^{-1}



Appendix 2 Plant species at each site in both years. [] = alien species.

Species	Site 1		Site 2		Site 3	
	2001	2002	2001	2001	2001	2002
[Acacia longifolia] (Andrews) Willd.			*	*	*	*
[Acacia melanoxylon] R.Br.	*	*				
[Acacia saligna] (Labill.) H.L.Wendl.			*	*	*	*
[Alstroemeria aurea] Graham						
Apatesia pillansii N.E.Br.		*				
Asparagus aethiopicus L.						
Asparagus declinatus L.	*					
[Avena barbata] Pott ex Link					*	
[Avena fatua] L.						
Berzelia lanuginosa (L.) Brongn.	*	*	*	*	*	
[Briza maxima] L.	*	*	*	*	*	*
[Briza minor] L.		*				
[Bromus diandrus] Roth		*				
[Bromus hordeaceus] L.		*				
Bromus leptoclados Nees	*					
Carpacoce spermacocea (Rchb.f.) Sond.	*	*				
Carpha glomerata (Thunb.) Nees			*	*	*	*
Carpobrotus edulis (L.) L.Bolus	*	*				*
Chenopodium album L.					*	
Chionanthus foveolatus (E.Mey.) Stearn						
Chrysanthemoides monilifera (L.) Norl. ssp. monilifera		*				
Chrysocoma coma-aurea L.			*	*		
Cliffortia dodecandra Weim.		*				
Cliffortia subsetacea (Eckl. & Zeyh.) Diels ex Bolus & Wolley-Dod		*				
[Conyza bonariensis] (L.) Cronquist				*		
[Conyza canadensis] (L.) Cronquist	*	*	*	*	*	*
Cotula turbinata L.		*				
Cotula vulgaris Levyns		*				
[Cucurbita moschata] (Duch. ex Lam.) Duch. ex Poir.	*					
Cunonia capensis L.	*	*				
Cuscuta nitida E.Mey. ex Choisy				*		*
[Cynodon dactylon] (L.) Pers.	*					
Cyperus species		*		*		*
Digitaria debilis (Desf.) Willd.	*	*		*	*	*
Disparago anomala Schltr. ex Levyns			*			
Droguetia ambigua Wedd.						
Ehrharta calycina Sm.			*		*	
Ehrharta erecta Lam.						*
Ehrharta ramosa (Thunb.) Thunb. ssp. aphylla (Schrad.) Gibbs-Russ.		*		*		
Ehrharta setacea Nees ssp. *uniflora* (Burch. ex Stapf) Gibbs-Russ.	*	*	*	*	*	*
Elegia thyrsifera (Rottb.) Pers.		*				
Erica caffra L.			*			
Erica hirtiflora Curtis			*	*		
Erica laeta Bartl.		*				
[Eucalyptus grandis] W.Hill ex Maiden			*	*		
Ficinia brevifolia Nees ex Kunth		*	*			

Species	**Site 1**		**Site 2**		**Site 3**	
	2001	**2002**	**2001**	**2002**	**2001**	**2002**
Ficinia bulbosa (L.) Nees		*				
Ficinia filiformis (Lam.) Schrad.	*			*		
Ficinia indica (Lam.) Pfeiff.		*	*	*		
Ficinia oligantha (Steud.) J.Raynal	*		*			
Ficinia secunda (Vahl) Kunth		*	*	*		
Ficinia species				*		
Ficinia tenuifolia Kunth			*			
Fuirena hirsuta (P.J.Bergius) P.L.Forbes	*		*	*	*	
[Fumaria muralis] Sond. ex W.D.J.Koch						
Helichrysum crispum (L.) D.Don			*			
Helichrysum cymosum (L.) D.Don	*	*	*	*	*	*
Helichrysum indicum (L.) Grierson		*				
Helichrysum litorale Bolus		*				
Helichrysum pandurifolium Schrank						
Hellmuthia membranacea (Thunb.) R.W.Haines & Lye	*		*	*		
Histiopteris incisa (Thunb.) J.Sm.	*	*	*	*	*	*
Indigofera capillaris Thunb.	*					
Ischyrolepis tenuissima (Kunth) H.P.Linder	*	*				
Isolepis ludwigii (Steud.) Kunth	*	*				
Isolepis marginata (Thunb.) A.Dietr.	*	*	*	*	*	*
Isolepis prolifer R.Br.	*	*	*	*	*	*
Isolepis tenuissima (Nees) Kunth	*					
Juncus capensis Thunb.	*	*	*	*	*	*
Juncus effusus L.		*		*		*
Kiggelaria africana L.						
[Lagurus ovatus] L.	*	*				
Lampranthus species		*				
[Lantana camara] L.					*	*
Laurembergia repens P.J.Bergius	*	*	*	*	*	*
Laurentia secunda (L.f.) Kuntze	*	*				
Leonotis leonurus (L.) R.Br.						*
Lobelia comosa L.			*	*		
Lobelia erinus L.				*		
Metalasia muraltiifolia DC.						
Metalasia muricata (L.) D.Don	*	*	*	*		
Micranthus alopecuroides (L.) Rothm.	*	*				
Oftia africana (L.) Bocq.		*	*	*		*
Otholobium parviflorum (E.Mey.) C.H.Stirt.			*			
Othonna parviflora P.J.Bergius			*	*		
Othonna quinquedentata Thunb.			*			
[Paraserianthes lophantha] (Willd.) I.C.Nielsen			*		*	
[Paspalum dilatatum] Poir.						*
[Paspalum urvillei] Steud.	*	*	*	*	*	*
Passerina vulgaris Thoday			*	*		
Pelargonium alchemilloides (L.) L'Hér.				*		
Pelargonium chamaedryfolium Jacq.	*		*			
Pelargonium cucullatum (L.) L'Hér. ssp. cucullatum	*	*	*	*	*	*
[Pennisetum clandestinum] Chiov.	*	*				*
Pennisetum glaucocladum Stapf & C.E.Hubb.						*

Species	Site 1		Site 2		Site 3	
	2001	2002	2001	2002	2001	2002
Pentaschistis airoides (Nees) Stapf	*		*	*		
Pentaschistis curvifolia (Schrad.) Stapf		*				
Pentaschistis glandulosa (Schrad.) H.P.Linder			*	*		*
Pentaschistis pallida (Thunb.) H.P.Linder		*				
[Persicaria serrulata] (Lag.) Webb & Moq.						
[Physalis peruviana] L.						
[Phytolacca americana] L.			*	*	*	*
[Pinus pinaster] Aiton	*	*				
Polypogon strictus Nees					*	*
[Populus x canescens] (Aiton) Sm.						
Prionium serratum (L.f.) Drège ex E.Mey.			*	*		
Prismatocarpus sessilis Eckl. ex A.DC.	*	*				
[Pseudognaphalium luteo-album] (L.) Hilliard & B.L.Burtt		*				
Pseudognaphalium undulatum (L.) Hilliard & B.L.Burtt				*		
Pseudoselago serrata (P.J.Bergius) Hilliard			*			
Psoralea pinnata L.	*	*	*	*		
Psoralea restioides Eckl. & Zeyh.		*				
Pteridium aquilinum (L.) Kuhn	*	*	*	*	*	*
Pycreus polystachyos (Rottb.) P.Beauv.	*	*		*		*
[Quercus robur] L.		*				
Rhus lucida L.	*	*	*	*		
[Ricinus communis] L.						
[Roella ciliata] L.			*			
Rorippa fluviatilis (E.Mey. ex Sond.) Thell. var. caledonica (Sond.) Marais						
[Rubus pinnatus] Willd.					*	*
[Rumex acetosella] L.		*			*	*
Senecio arenarius Thunb.				*		
Senecio burchellii DC.				*	*	*
Senecio crassiusculus DC.					*	
Senecio elegans L.						
Senecio pinnulatus Thunb.		*	*			
Senecio pterophorus DC.						
Senecio pubigerus DC.	*	*	*	*	*	*
Senecio rigidus L.	*	*	*	*	*	*
[Solanum nigrum] L.	*	*	*	*		*
[Sonchus oleraceus] L.			*			
Sporobolus africanus (Poir.) Robyns & Tournay	*	*	*	*		
Stoebe cinerea (L.) Thunb.						
Stoebe fusca (L.) Thunb.		*				
Syncarpha vestita (L.) B.Nord.		*				
[Taraxacum officinale] Weber sensu lato	*	*		*	*	*
Tetraria capillacea (Thunb.) C.B.Clarke			*			
Tetraria exilis Levyns		*				
Themeda triandra Forssk.				*		
Todea barbara (L.) T.Moore	*	*	*	*		
Trachyandra divaricata (Jacq.) Kunth				*		
Ursinia anthemoides (L.) Poir.			*			
Ursinia tenuifolia (L.) Poir.		*				
Vellereophyton dealbatum (Thunb.) Hilliard & B.L.Burtt			*			

Species	Site 1		Site 2		Site 3	
	2001	2002	2001	2002	2001	2002
Virgilia oroboides (P.J.Bergius) Salter	*	*				
Wahlenbergia parvifolia (P.J.Bergius) Lammers			*	*		
Zantedeschia aethiopica (L.) Spreng.	*	*				

Appendix 3 **Macroinvertebrate abundances (numbers of individuals/m^2) in both years.** Acronyms as per Table 4.2. 1/2 = year 1/2.

Site1

	1TrCoGr	1BfGrSa	1SsCoGr	1SsGrSaMv	1MrSa	1MrCoSa	2StCoSaMv	2StCoSa	2StSa	2StBr
Porifera-sponges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nematoda	6.3	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
Collembola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrozoa-Hydra sp	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planariidae-Planaria	131.3	0.0	72.0	14.7	20.0	10.5	3.0	0.0	0.0	0.0
Lumbiculidae	125.0	0.0	80.0	71.3	174.0	32.5	60.0	13.0	6.0	2.0
Naididae	0.0	16.0	1.0	1.3	28.0	0.0	0.0	1.0	0.0	2.0
Hirudinea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphipoda	5625.0	880.0	800.0	1066.7	3132.0	1050.0	1600.0	450.0	800.0	66.0
Brachyura-Decapoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natantia-shrimps	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cladocera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Copepeoda	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrachnellae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oribatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonemouridae	12.5	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0
Perlidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polymitarcyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemeridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oligoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heptageniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptophlebiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemerellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tricorythidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Prosopistomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Protoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Platycnemidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenagrionidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Calopterygidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorocyphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Libelullidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1TrCoGr	1BfGrSa	1SsCoGr	1SsGrSaMv	1MrSa	1MrCoSa	2StCoSaMv	2StCoSa	2StSa	2StBr
Gomphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aeshnidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corduliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Zygoptera juvs.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naucoridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nepidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Belastomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corixidae	0.0	0.0	0.0	0.0	4.0	0.0	0.0	0.0	0.0	0.0
Gerridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Veliidae	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
Corydalidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 1 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 2 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 3 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 4 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- >5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Philopotamidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psychomyiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ecnomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydropsychidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptoceridae	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.5	2.0
Pyraustidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dytiscidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Elmidae	131.3	8.0	17.0	8.0	28.0	4.5	9.0	1.5	0.5	0.0
Elmidae (A)	168.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dryopidae (A)	6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrinidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrinidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Haliplidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Helodidae	0.0	0.0	0.0	3.3	0.0	3.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae (A)	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Limnichidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1TrCoGr	1BfGrSa	1SsCoGr	1SsGrSaMv	1MrSa	1MrCoSa	2StCoSaMv	2StCoSa	2StSa	2StBr
Psephenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Torridincolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Noteridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Blephariceridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Psychodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Culicidae	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dixidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Simulidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	6.3	300.0	71.0	0.0	24.0	13.5	2.0	15.0	1.5	4.0
Ceratopgonidae	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
Tabanidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Athericidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Empedidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephydridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Muscidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lymnaeidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Malaniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planorbidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aneylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corbiculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Spahaeriidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unionidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Site 2, year 1

	1SsBoGrSa	1BfSa	1MrMv	1MrBoSa	1SsMvBoSa	1MrBrSa	1BfBrSa	1MrGrSa	1MrCo	1MrBoGr	1MrMvCoSa	1BfMvSa
Porifera-sponges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nematoda	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Collembola	0.0	0.0	0.0	0.0	0.7	0.0	0.0	4.0	0.0	0.0	0.0	0.0
Hydrozoa-Hydra sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planariidae-Planaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lumbiculidae	2.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	2.0	0.2	2.5	0.7
Naididae	0.0	0.1	0.3	0.0	0.0	1.0	0.2	4.0	2.0	0.3	0.7	0.0
Hirudinea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphipoda	20.0	0.0	0.0	2.0	2.0	102.0	0.2	4.0	10.0	5.8	6.1	4.0
Brachyura-Decapoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natantia-shrimps	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cladocera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Copepeoda	0.0	0.0	0.0	0.0	2.0	0.0	0.0	2.0	0.0	0.0	0.1	0.0
Hydrachnellae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.3	0.0
Oribatidae	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Notonemouridae	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
Perlidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polymitarcyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemeridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	1.0	0.0	5.0	32.0	0.0	1410.0	0.0	4.0	204.0	20.2	14.0	1.3
Oligoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heptageniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptophlebiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemerellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tricorythidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Prosopistomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Protoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Platycnemidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenagrionidae	1.0	0.0	2.0	0.0	9.3	0.0	1.3	20.0	0.0	0.7	0.8	5.3
Calopterygidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorocyphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Libelullidae	15.0	3.8	1.3	0.5	2.0	0.0	2.7	12.0	0.0	0.2	0.3	13.3
Gomphidae	22.0	0.4	1.0	3.0	0.7	63.0	2.2	18.0	4.0	0.7	4.4	28.0
Aeshnidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.2	0.8	0.0
Corduliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1SsBoGrSa	1BfSa	1MrMv	1MrBoSa	1SsMvBoSa	1MrBrSa	1BfBrSa	1MrGrSa	1MrCo	1MrBoGr	1MrMvCoSa	1BfMvSa
Zygoptera juvs.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naucoridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nepidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Belastomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corixidae	1.0	0.4	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
Gerridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Veliidae	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corydalidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 1 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 2 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 3 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 4 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- >5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Philopotamidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psychomyiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ecnomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae	0.0	0.0	2.8	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.1	0.0
Hydropsychidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptoceridae	2.0	0.1	0.5	0.5	13.3	1.0	1.3	2.0	2.0	1.0	0.5	1.3
Pyraustidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dytiscidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Elmidae	26.0	0.3	0.0	3.0	0.0	2.0	1.5	0.0	2.0	1.2	10.0	0.7
Elmidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dryopidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Gyrinidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrinidae (A)	1.0	0.0	0.0	0.5	2.7	0.0	0.0	0.0	0.0	0.0	0.0	2.0
Haliplidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Helodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	1.1	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae (A)	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Limnichidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Psephenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Torridincolidae	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Noteridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1SsBoGrSa	1BfSa	1MrMv	1MrBoSa	1SsMvBoSa	1MrBrSa	1BfBrSa	1MrGrSa	1MrCo	1MrBoGr	1MrMvCoSa	1BfMvSa
Blephariceridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.1	0.0
Psychodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dixidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Simulidae	1.0	0.0	0.0	4.5	0.7	5.0	0.0	0.0	52.0	20.0	1.2	1.3
Chironomidae	71.0	5.7	16.0	54.0	16.7	55.0	20.5	62.0	190.0	12.2	35.5	10.0
Ceratopgonidae	0.0	0.0	0.0	0.5	0.0	4.0	0.0	2.0	0.0	1.2	0.3	0.0
Tabanidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Athericidae	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.0	1.2	9.5	0.0
Empedidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephydridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stratiomyidae	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Muscidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lymnaeidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Malaniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planorbidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aneylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corbiculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Spahaeriidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unionidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Site 2, year 2

	2BfSa	2StMvSa	2MrSa	2MrBr	2CaAlBr	2StBrSa	2BfCoSa	2SsSa	2SsBoSa	2MrBoCoGr	2CaBoCo
Porifera-sponges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nematoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Collembola	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrozoa-Hydra sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planariidae-Planaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lumbiculidae	0.0	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naididae	0.0	2.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hirudinea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphipoda	1.0	0.0	178.0	6.7	0.0	0.0	4.0	32.0	16.0	1.7	90.0
Brachyura-Decapoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natantia-shrimps	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cladocera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Copepeoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrachnellae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oribatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonemouridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Perlidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polymitarcyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemeridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	0.0	0.0	12.0	1466.7	380.0	0.0	16.0	24.0	12.0	13.3	770.0
Oligoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heptageniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptophlebiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemerellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tricorythidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Prosopistomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Protoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Platycnemidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenagrionidae	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Calopterygidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorocyphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Libelullidae	0.0	18.0	2.0	0.0	0.0	0.5	0.0	0.0	0.0	1.7	0.0
Gomphidae	12.0	2.0	30.0	3.3	0.0	0.8	62.0	20.0	10.0	16.7	33.3
Aeshnidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corduliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	2BfSa	2StMvSa	2MrSa	2MrBr	2CaAlBr	2StBrSa	2BfCoSa	2SsSa	2SsBoSa	2MrBoCoGr	2CaBoCo
Zygoptera juvs.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naucoridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nepidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Belastomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corixidae	0.0	30.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gerridae	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Veliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corydalidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 1 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 2 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 3 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 4 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- >5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Philopotamidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psychomyiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ecnomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydropsychidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptoceridae	1.0	0.0	4.0	0.0	0.0	1.5	12.0	8.0	4.0	23.3	0.0
Pyraustidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dytiscidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Elmidae	0.0	0.0	0.0	26.7	60.0	0.3	2.0	4.0	2.0	3.3	70.0
Elmidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dryopidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrinidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gyrinidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Haliplidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Helodidae	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Limnichidae	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psephenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Torridincolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Noteridae	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	2BfSa	2StMvSa	2MrSa	2MrBr	2CaAlBr	2StBrSa	2BfCoSa	2SsSa	2SsBoSa	2MrBoCoGr	2CaBoCo
Blephariceridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psychodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dixidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Simulidae	0.0	0.0	0.0	3430.0	1430.0	1.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	4.0	32.0	62.0	280.0	320.0	1.0	6.0	28.0	14.0	85.0	46.7
Ceratopgonidae	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	0.0	1.7	6.7
Tabanidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Athericidae	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7
Empedidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephydridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Muscidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lymnaeidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Malaniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planorbidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aneylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corbiculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Spahaeriidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unionidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Site 3

	1BfMv	1BfBrGrSa	1MrIvCoGr	1CaBo	1BfSa	1BfBo	1MrBr	2MrBoCoGr	2BfBoSa	2BfMv	2CaBo	2BfSa	2StGrSa	2MrBr
Porifera-sponges	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nematoda	0.0	0.0	20.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	2.0	0.0	5.0
Collembola	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrozoa-Hydra sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planariidae-Planaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lumbiculidae	0.0	0.0	15.0	0.0	0.0	0.0	0.0	7.3	0.0	1.0	2.0	1.0	0.0	0.0
Naididae	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	4.0	0.0	0.0	0.0
Hirudinea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphipoda	6.7	60.0	40.0	25.0	0.2	0.0	0.3	24.0	6.0	9.0	0.0	1.0	44.0	1.7
Brachyura-Decapoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Natantia-shrimps	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cladocera	0.0	60.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.0	0.0
Copepeoda	0.0	0.0	10.0	0.0	0.2	0.7	0.0	0.0	0.0	0.0	0.0	4.0	4.0	0.0
Hydrachnellae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	4.0	0.0
Oribatidae	3.3	0.0	20.0	0.0	0.0	0.7	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonemouridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	14.0	0.0	0.0
Perlidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polymitarcyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemeridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	0.0	0.0	40.0	300.0	0.0	0.7	6.1	92.7	4.0	2.0	176.0	1.0	0.0	1288.3
Oligoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heptageniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptophlebiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemerellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tricorythidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Prosopistomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lestidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Protoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Platycnemidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenagrionidae	50.0	30.0	110.0	25.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	0.0	0.0
Calopterygidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorocyphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Libelullidae	56.7	570.0	35.0	0.0	13.3	18.0	0.7	1.3	102.0	14.0	0.0	61.0	244.0	1.7
Gomphidae	0.0	440.0	45.0	0.0	8.2	0.3	0.5	4.0	72.0	1.0	0.0	71.0	188.0	15.0
Aeshnidae	0.0	0.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corduliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1BfMv	1BfBrGrSa	1MrIvCoGr	1CaBo	1BfSa	1BfBo	1MrBr	2MrBoCoGr	2BfBoSa	2BfMv	2CaBo	2BfSa	2StGrSa	2MrBr
Zygoptera juvs.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naucoridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nepidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Belastomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corixidae	0.0	0.0	0.0	0.0	0.5	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gerridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Veliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corydalidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 1 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 2 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 3 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 4 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- 5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cased caddis- >5 type	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Philopotamidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Psychomyiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ecnomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae	6.7	0.0	20.0	50.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydropsychidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptoceridae	23.3	70.0	15.0	0.0	0.5	0.7	1.1	0.0	10.0	13.0	0.0	7.0	16.0	0.0
Pyraustidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7
Dytiscidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Elmidae	0.0	40.0	0.0	25.0	4.3	0.0	0.6	2.0	14.0	0.0	0.0	4.0	8.0	8.3
Elmidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Dryopidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
Gyrinidae	0.0	0.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	3.0	2.0	0.0	0.0	0.0
Gyrinidae (A)	0.0	0.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Haliplidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Helodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
Hydraenidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae (A)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
Limnichidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	12.0	1.0	0.0	0.0	0.0	0.0
Psephenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Torridincolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Noteridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	1BfMv	1BfBrGrSa	1MrIvCoGr	1CaBo	1BfSa	1BfBo	1MrBr	2MrBoCoGr	2BfBoSa	2BfMv	2CaBo	2BfSa	2StGrSa	2MrBr
Blephariceridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Psychodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dixidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Simulidae	0.0	0.0	30.0	2825.0	0.0	0.7	0.6	1.3	0.0	0.0	102.0	1.0	0.0	11.7
Chironomidae	156.7	810.0	385.0	3625.0	13.8	91.3	11.8	60.0	70.0	20.0	88.0	10.0	32.0	378.3
Ceratopgonidae	0.0	10.0	10.0	25.0	0.2	0.0	0.0	2.0	16.0	0.0	4.0	0.0	4.0	1.7
Tabanidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Athericidae	0.0	30.0	30.0	25.0	0.5	0.0	0.5	22.7	2.0	3.0	0.0	0.0	0.0	71.7
Empedidae	0.0	0.0	0.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephydridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Muscidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lymnaeidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Malaniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planorbidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aneylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corbiculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Spahaeriidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unionidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0