Evaluation of macro-invertebrates as bio-indicators of water quality and the assessment of the impact of the Klein Plaas dam on the Eerste River

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Declaration

I, the undersigned, hereby declare that the work contained in this thesis is my own original work and that I have not previously in its entirety or in part submitted it at any university for a degree.

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Date: ………………….……………..

Abstract

A semi-arid country, like South Africa, with unpredictable seasonal rainfall, is subject to great scarcity in water and an ever-increasing demand from the rising human population. Therefore, efficient reservoirs as well as monitoring methods are needed to manage the South African water supply.

This study was undertaken on the Eerste River in the Western Cape, South Africa, focusing on the impact of the Klein Plaas dam system on the benthic macroinvertebrates. The study also examined the use of benthic macroinvertebrates as bioindicators of water quality with special reference to the South African Scoring System Version 5(SASS5) that is currently being used nationally.

The impoundment of the water, as well as the inter-basin transfer programme and the experimental cage-culture trout farm, all play a significant role in the disturbance impact of the dam on the Eerste River system. The disturbance is manifested as a drop in water quality that can be seen in the distribution of keystone species, changes in the riparian vegetation, as well as in physical-, chemical-, and biomonitoring evaluations.

The study also indicated that the SASS5 is effective, but needs some adjustments, such as inclusion of a prediction phase, finer spatial-scale methodologies and greater consideration of the rarity of species.

Opsomming

Suid-Afrika as `n semi-aride land, met ‘n onvoorspelbare reënval, is onderhewig aan `n gedurige water tekort en weens die stygende bevolkingsgetalle is daar ‘n gepaardgaande aanvraag vir water. Dus is effektiewe bewaring en moniteringsmetodes nodig vir die bestuur van Suid-Afrika se watervoorraad.

Die studie is in die Eersterivier, gelee in die Wes Kaap, Suid-Afrika, uit gevoer met die fokus op die impak van die Klein Plaas dam sisteem, op die bentiese makroinvertebrate. Die studie dek ook die gebruik van bentiese makroinvertebrate as bioindikatore van waterkwaliteit en spesifiek in die gebruik van die “South African Scoring System Version 5”(SASS5).

Die opdamming van die water, sowel as die oordragprogram van water vanaf ander riviere en die eksperimentele visteelstasie speel `n belangrike rol in die versteuringseffek van die dam op die Eersterivier sisteem. Die versteuring wys op ‘n verlaging van waterkwaliteit, wat sigbaar is in die distribusie van belangrike spesies, veranderinge in die rivierbank se plantegroei, sowel as die fisiese, chemiese en biomonitoriese evaluasies.

Verder toon die studie dat die SASS5 effektief toegepas kan word, maar aanpassings soos die insluiting van ‘n voorspellingsfase, die opbreek en spesialisering gebaseer op kleiner geografiese areas, sowel as om die skaarsheid van `n spesie in ag te neem, benodig word.

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Chapter One

INTRODUCTION

Streams and rivers form the most widespread type of surface freshwater habitat in the world (Zwick, 1992). Being central to the global water cycle, they interconnect all of its components, fresh as well as saline. Although at any given moment, the amount of water present in a river may be relatively small, a very large proportion of the freshwater in the system passes thorough any point along the stream over time.

Rivers irrigate and drain catchments, the latter largely shaped by the action of its flow. In contrast, the quality of the catchment area also affects the rivers. In turn, the conditions in the river affect both the quantity and quality of our most vital resource, clean fresh water (Zwick, 1992).

As rivers are on the receiving end of the drainage system of any catchment area, they are highly vulnerable to change in land use and other human activities. Their flow is manipulated to provide water supplies. Barriers are constructed for flood control, gabions and walls are used to counteract erosion, while river channels and canalisation are used as conduits for delivery of irrigation water and disposal of wastes. These practices have brought many benefits to society, but they have also resulted in widespread degradation of the river ecosystems (King & Schael, 2001).

During later half of the twentieth century there was tremendous industrial development leading to the pollution of water, air, soil and general apathy (Swaminathan, 2003). The main concerns from the increasing industrial and vehicular emissions, are those regarding acid rain, ozone layer depletion, greenhouse gases and global warming, with most of these problems not only being environmental problems of their respective countries, but they are also issues that are transboundary and truly an international problems (Swaminathan, 2003). However, the greatest concern of all is the influence of pollution on our freshwater regime.

With the need to monitor water quality, there has been a proliferation of techniques for rapid bioassessment of rivers and evaluation of water quality (Reviewed by Rosenberg & Resh, 1993; Metcalf-Smith, 1994; Resh, 1995; Dickens & Graham, 2002).

These techniques are used for health assessment of general river conditions as influenced by a variety of factors, especially water quality. Many of these methods have been applied by regulatory authorities regarding bioassessment data as valuable for the management of aquatic resources (Dickens & Graham, 2002).

The United Nations, being concerned with the quality of the world's freshwater supply, convened the Convention on Climate Change in Kyoto, Japan to halt or reduce the negative effects of environmental degration of this resource (Swaminathan, 2003). This awareness has resulted in a demand for more information on monitoring the environment, so that we can best preserve our freshwater resources for future generations.

South Africa, as a semi-arid country, with unpredictable seasonal rainfall, is subject to great scarcity in water and an ever-increasing human population, and it is expected to increase its water demands beyond supply within the next two decades (Basson *et al.*, 1997). Furthermore, South Africa is significant for biodiversity, as two of the world's 25 biodiversity hotspots are within the country (Myers *et al.*, 2004). This puts great pressure on evaluation and management of riverine ecosystems.

Biodiversity is used as a tool for assessing, prediction and transformation landscape structure, making it a valid component of policies applicable to rural, industrial and urbanized areas so as to decrease human mismanagement and to lessen pollution levels (Wilson, 1997). The importance of biodiversity in directing environmental policy, presupposes that organisms and their complex interactions respond favourably to human landscape management and impacts in different ways, with some organisms responding quicker and more definitively than others (Paoletti, 1999).

Need has created an urgency to develop monitoring methods that can indicate the ecological status of riverine systems as they respond to natural and biotic activities, such monitoring feed directly into practicable conservation strategies. Monitoring methods for ecosystems can be classified as physical, chemical or biological. Biological monitoring (biomonitoring) makes use of the living components of the studied environment, and indicates, as well as assesses, ecological degradation, transformation, improvement or other effects, due to a certain cause at a specific or at similar locations, with minimal use of equipment in the field and which non-specialists can do. The basic aim of biomonitoring is to indicate the emerging catastrophes at an early stage. After initial standardization and establishment, the biomonitoring techniques must also be cost effective and become part of the people's mindset in pinpointing environmental degradation to serve as an effective warning system (Swaminathan, 2003).

Benthic macroinvertebrates being widespread and sensitive to environmental changes (Saifutdinova & Shangaraeva, 1997), are the group of organisms most offen used for assessment of fresh water quality (Resh, 1995). In South Africa, macroinvertebrate bioassessments are undertaken on the basis of a modification of the methodology used by the British Monitoring Working Party (BMWP). This modified system is the South African Scoring System and is currently in its fifth version (SASS5) (Dickens & Graham, 2002). Application of such bioindicators can be used to improve the environment and to augment awareness of the living creatures to obtain better appreciation of their crucial role in sustaining life on the planet (Paoletti, 1999).

This study was undertaken on the Eerste River, arising in the Hottentots Holland Catchment area in the Western Cape, South Africa. The water quality of the sampling area upstream from the Klein Plaas dam (also mentioned in some literature as the Jonkershoek dam) of the Eerste River has remained relatively undisturbed (Brown & Dallas 1995). In the sampling area downstream from the dam, the water quality has, however, been affected by agricultural runoff, as well as an increase in detrimental activities in the catchment itself. In addition, the construction of the Klein Plaas dam and the other disturbances in the natural flow regime, especially the extraction of water for residential and irrigation purposes, have in all probability, resulted in an increase in the rate of decline in water quality (Brown & Dallas, 1995).

Against this background, the aims of this study were:

1. To make an ecological assessment on the influence of the Klein Plaas dam on the Eerste River ecosystem, with special reference to macroinvertebrates, and

2. To evaluate SASS5 using macroinvertebrates as indicators of water quality in this riverine ecosystem.

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Chapter Two

LITERATURE REVIEW

Five distinct, but pertinent, topics for assessing the Klein Plaas dam as well as the biomonitoring of the Eerste River are researched. These are:

- The Eerste River itself,
- A river ecosystem,
- Bioindicators of water quality,
- South African Scoring System, and
- Aquatic macroinvertebrates.

2.1 EERSTE RIVER

2.1.1 The river course

The Eerste River (Fig. 2.1.1) rises in the Dwarsberg to the southeast of Stellenbosch, and flows in a northwesterly direction through the town and then southerly towards False Bay near Macassar, where it ends in a small estuary, stretching over approximately 40 km (Brown & Dallas 1995). It has a catchment area of 420 km^2, and a mean annual runoff (MAR) that is variable fluctuating between 48 x 10^6 to 167 x 10^6 $m^3.a^{-1}$ (Brown & Dallas 1995). At and near its source, the river flows through the Jonkershoek valley that forms part of the Hottentots-Holland Nature Reserve. It is in this area where the Klein Plaas dam is situated in the system (33°58'42.0" S; 18°56'36.0" E). The Lang River and Swartboskloof River enters the Eerste River in this mountain stream zone.

From the dam, the river flows out of the reserve past Cape Nature’s Jonkerhoek Station through Assegaaibos Nature Reserve and down to Stellenbosch, receiving many smaller tributaries on route. In Stellenbosch, the river is canalised along much of its length, with parts of the river being divided into numerous street canals forming part of the historical irrigation system.

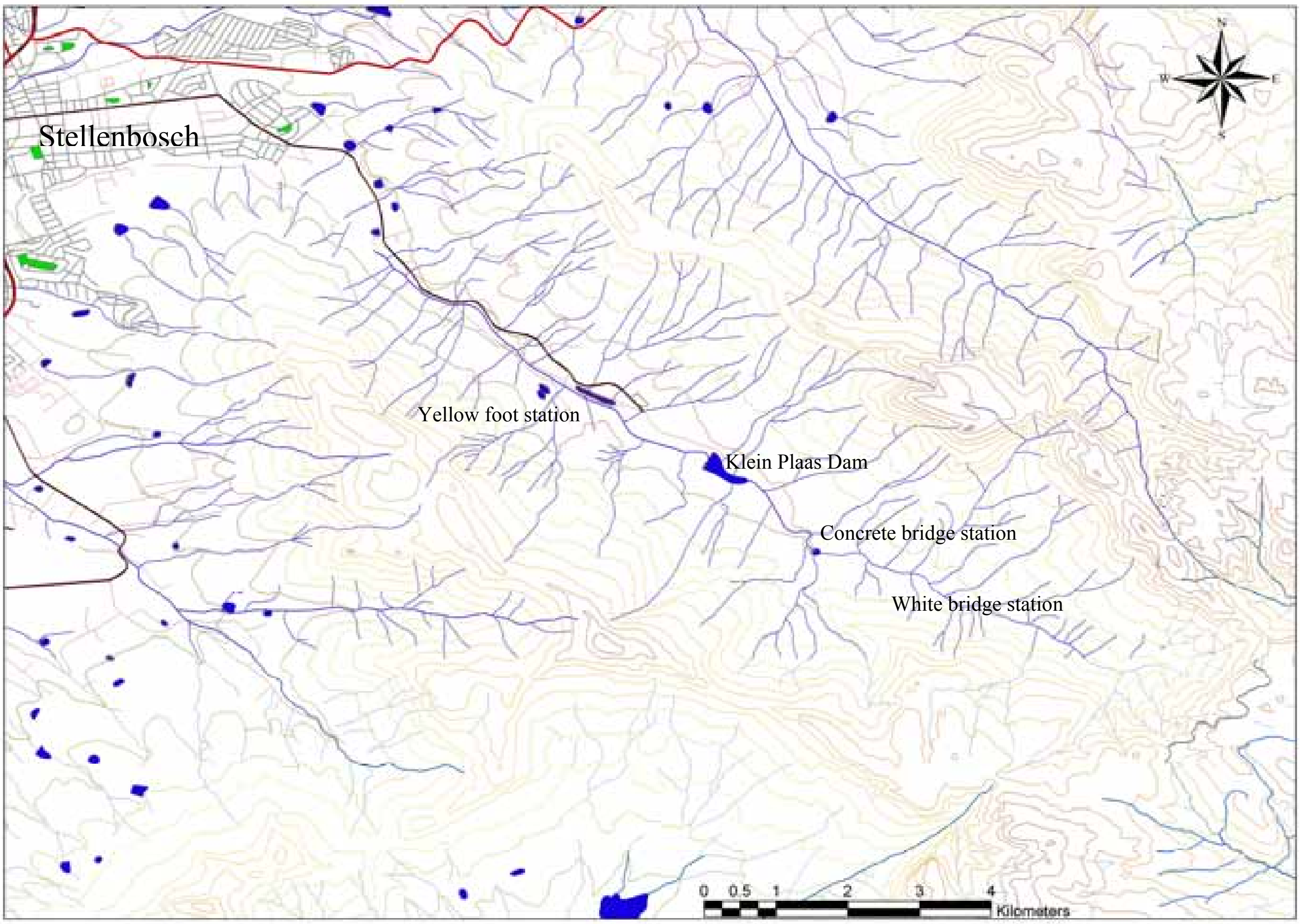

Figure 2.1.1 Eerste River, Jonkershoek

Immediately downstream of Stellenbosch, the Eerste River is joined by its first major tributary, the Plankenbrug River. The river then flows past Stellenbosch Farmer's Winery before meeting the Veldwagters River. From this point onwards, the Eerste River flows through vineyards and agricultural land to the sea, being joined near Vlottenberg, by the Sanddrif and the Blouklip Rivers, and near Uitsig, by the Bonte River. Immediately before entering at False Bay, the Eerste River joins the Kuils and Helderberg Rivers (Brown & Dallas 1995).

2.1.2 **Zonation of the Eerste River**

The Eerste River consists of three zones:

1. The mountain-stream zone is the 7 km zone situated within the Hottentots Holland Nature Reserve.
2. The upper-river zone, a 5 km stretch from the boundary of the reserve to the outskirts of Stellenbosch.
3. The lower-river zone that flows towards Macassar and into the sea.

The mountain-stream zone starts at the source of the river in the Jonkershoek valley, with a stream width of 5-7 m, and an average gradient of 24 $m.km^{-1}$ (Brown & Dallas 1995). The substratum consists of boulders, large stones and bedrock. The upper-river zone has a stream width of 7-11 m and the average gradient is 12 $m.km^{-1}$ (Brown & Dallas 1995). The substratum is similar to that of the mountain-stream zone. The lower-river zone is a 28 km stretch of river that widens to 8-18 m with an average gradient of 2 $m.km^{-1}$ (Brown & Dallas 1995). Its substratum consists of stones and pebbles on coarse sand.

2.1.3 Human interference in the system

Just upstream of the border of the nature reserve, the river is impounded by the Kleinplaas Dam, which was built in 1981, and has a storage capacity of 377 000 m^3. At this point the river also receives water from the Riviersonderend-Berg-Eerste Government Water Scheme (RBEGS). The Klein Plaas Dam is also the site of an experimental cage-culture trout farm run with the support of the Department of Genetics (Aquaculture) of the University of Stellenbosch.

Slightly upstream of the Klein Plaas dam are two weirs, one a weir that during the summer diverts almost all the Eerste River flow in the direction of Ida's Valley (Brown & Dallas 1995). Furthermore, pine plantations are upslope in this area.

In the area beyond the Jonkershoek Nature reserve the riparian land-use is primarily vineyards, although there are also recreational and residential areas adjoining the river as well as an experimental aquacultural facility (also run by University of Stellenbosch) associated with the CapeNature holdings.

Water is being abstracted by the Stellenbosch Municipality to supply the town of Stellenbosch, as well as riparian landowners along much of its length of the river.

Treated effluent enters from the Stellenbosch Municipality Sewage works the system via the Veldwagters River. Furthermore the Plankenbrug River infuses polluted water from the Cloetesville informal settlements and Stellenbosch industrial area.

2.1.4 The flow

The flow of the Eerste River is distinctly seasonal and dependent on the annual precipitation. Jonkershoek is regarded as one of South Africa's areas with a very high precipitation levels, significantly higher than the average of rainfall per annum of 500 mm for the whole of South Africa (Madikizela, 2001). The low-flow period in the Eerste River extends from about November through to the end of March. Flow is, on average, lowest in January and February, although water abstraction ensures that flows in the river remain unnaturally low throughout the summer months. Being in the Winter Rainfall area, the water level is at its peak during the June to September periods. The area upstream from the Klein plaas increases in the amount of rapids as well as in the strength of the current. Downstream from the dam, the river drastically increases in water volume, making accessibility of sites by foot impossible.

2.1.5 Riparian vegetation

The mountain-stream zone, upstream of the diversion weir and Kleinplaas Dam, is assigned a high conservation status, generally unspoiled, and much of the riparian belt consists of indigenous fynbos. Although the area just below the dam has extensive pine plantations, the riparian belt has, for the most part, been left undisturbed.

The upper-river zone is impacted upon by agricultural activities as well as the removal of indigenous riparian vegetation, but is still considered to be largely natural with few disturbances. The riparian vegetation reflects the land-use of the area, and in most places is dominated by aliens plants such as oaks, *Quercus robur*, white poplar, *Populus canescens*, Port Jackson, *Acacia saligna, Eucalyptus roman* and black wattle, *Acacia mearnsii*. However, in places, the indigenous riparian belt remains intact, represented by species such as the smalblaar, *Metrosideros angustifolia,* and wild almond, *Brabejum stellatifolium* (Joffe, 2003).

Once the river enters Stellenbosch, the indigenous riparian vegetation is almost non-existent, while riparian vegetation for the river exiting Stellenbosch consist predominantly of the invasive alien, *Acacia mearnsii* (Brown & Dallas 1995).

2.1.6 Related studies on the Eerste River system:

- Surveys of water quality in the mid-1960s (Steer, 1964, 1965, 1966).
- Catchment studies (Van der Zel & Kruger, 1975; Scott & van Wyk, 1990; Versveld, 1993).
- Studies of the small mountain streams draining catchments in the Jonkershoek Valley (Davies *et al*., 1987; Stewart & Davies, 1990; Prochaska *et al*., 1992).
- An Eerste River Catchment Management Report (Petitjean, 1987).
- A study of the degradation of the Eerste River from a legal perspective (Wiseman & Simpson, 1989).
- Western Cape Systems Analysis (Ninham Shand Consulting Engineers, 1994).
- Schools Water Project (Boucher & Schreuder, 1993).
- SASS4 evaluation of the Eerste River (Dallas *et al*., 1995).
- Situation assessment of the riverine ecosystem (Brown & Dallas, 1995).

2.2 THE RIVER ECOSYSTEM

A watershed begins with small trickles of water, probably arising from precipitation, which gather and consecutively form larger stream types and eventually rivers. Over evolutionary time, rivers are capable of carving out the earth to better suit their pathway, carrying not only water to the ocean, but transferring vast amounts of organic matter and nutrients downstream. This organic matter derives from seepage, leaf drop, erosion, decay of creatures and other debris that drops into the system.

Vannote *et al*. (1980) emphasized that a river is more than the sum of its parts, and thus they developed the principle of the River Continuum Concept (RCC), suggesting that all rivers possess continuous gradients of physical and chemical continuum, not only in space but also in time. The RCC details the role of driving variables in rivers and the responses they bring forth in the biota. Each species of riverine organism will be confined to those parts of the river, when physical and chemical conditions are suitable for their requirements (Davies & Day, 1998; Vannotte, 1980). To understand what is happening at any point along the continuum, one must understand both what is happening upstream and what is entering from the watershed.

2.2.1 River Continuum Concept

The River Continuum Concept (RCC) (Fig. 2.2.1) describes the physical processes (geology, climate) outside of a river affecting the biological processes (vegetation) along a river, which in turn, affects the physical and biological processes within a river (temperature, nutrients). In simple terms, the RCC views all rivers as possessing continuous gradients of physical and chemical conditions that are progressively and continuously modified downstream from the headwaters to the sea.

Usually, the upstream to downstream reaches are described in terms of their stream order. This is the position of a stream in the hierarchy of tributaries. Second order streams have only first order streams as tributaries; third order streams are formed when two, second order streams join, etc. The RCC is divided into three zones along the continuum: the upper reach (orders 1-3), the mid-reach (orders 4-5) and the lower reach (order 6 and up). Table 2.2.1 shows the different characteristics of the different regions.

The physical basis of the RCC is size and location along the gradient from a tiny spring brook to a large river. Along its length, the stream increases in size, gathers tributaries and drains an increasingly larger catchment area. Stream order, discharge and watershed area have each been advocated as the physical measure of position along the river continuum. Stream order enables visualization of this concept, and for this reason is the most widely used (Allan, 1995).

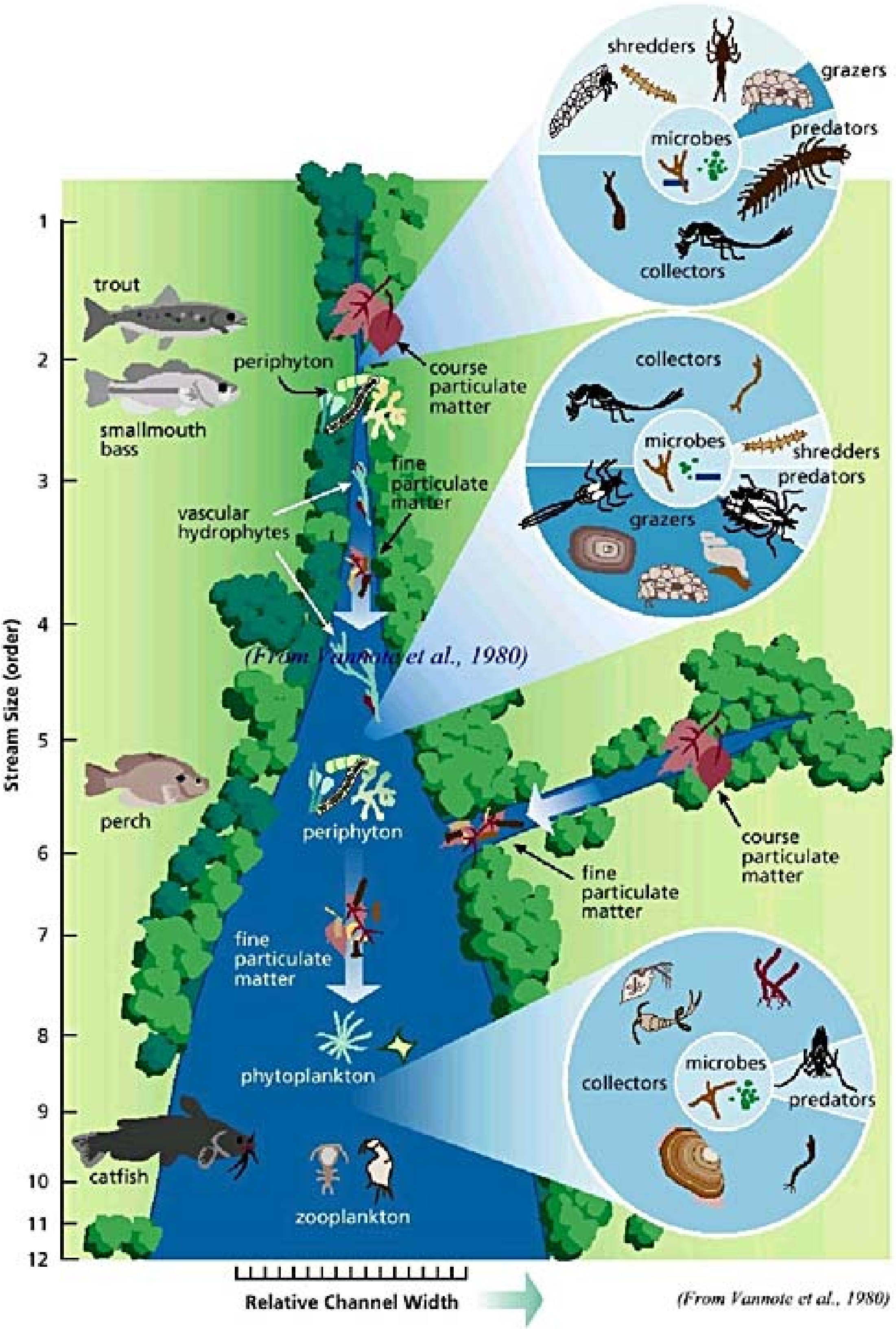

Figure 2.2.1 Illustration of the River Continuum Concept.

Table 2.2.1. Summary of River Continuum Concept's Characteristics

Characteristic	**Upper Reach**	**Mid-Reach**	**Lower Reach**
Light Penetration	Low	High	Low
Water Clarity	High	High	Low
Temperature	Low	Moderate	High
Current	Varied	Varied	High
Shading	High	Moderate	Low
Bottom Composition	Rock	Cobble/Gravel	Sand/silt
Habitat Diversity	Low	High	Low
Habitat Types	Fall/pool	Riffle/pool	Run
Width	Low	Moderate	High
Depth	Low	Varied	High
Dissolved Gases	High	Moderate	Low
Major Ions	Low	Moderate	High
Nutrients	Low	Moderate	High
Dominant Food Type	CPOM/FPOM	Periphyton	Phytoplankton
Dominant Feeding Group	Shredders Collectors	Grazers Collectors	Collectors
Plants	Attached Mosses	Attached Periphyton	Floating Phytoplankton

There is a pattern of progressive physical change that occurs from higher in the watershed to the base. The changes include:

- an increase in stream size, width, depth, velocity, and flow volume;
- a decrease in shading (canopy cover) by the riparian zone;
- a decrease in size of streambed substrate; from boulders in mountain streams to sandy-bottomed rivers.

The second important feature of the RCC deals with the response of the biota or riverine communities to the conditions in which they live. In particular, it deals with the supply of organic energy or food for the biota and the manner in which the biota produces and utilises this food. In other words, it describes the form of, and changes in the balance between allochthonous (externally-generated) and autochthonous (internally-generated) food resources as they change from headwaters to the sea. An allochthonous system is heterotrophic, consuming more organic material than it produces, while an autochthonous system is autotrophic, usually producing more organic material than it consumes (Davies & Day, 1998).

Each species of riverine organism will be confined to those parts of the river where physical and chemical conditions are suitable for it. Those with similar requirements will thus form species assemblages characteristic of particular reaches of the river. In turn, the organisms themselves alter the conditions that prevail further downstream. Thus, whatever happens in upstream reaches (a leaf being eaten, chemicals being leached from silt, the death of an animal or plant) will influence downstream processes such as decomposition and nutrient cycling, and will also influence the communities of organisms downstream. In this way, there is a continuous gradation along the length of any river, with the gradients of physical and chemical conditions eliciting a series of biological responses (Davies & Day, 1998).

The RCC states that producer and consumer communities become established in harmony with the dynamic physical conditions that include width, depth, velocity, flow volume, and temperature of the river (Vannote *et al*. 1980). For example, as the size of a river increases from a headwater stream to a mid-sized river, the influence of the surrounding riparian forest decreases due to the change in the dominant biological community. Riparian vegetation may be the single most important component to headwater stream stability, production and diversity (Rosgen, 1985).

To understand the different functional feeding groups found in the RCC, the food and energy source of these groups are discussed.

2.2.2 Particle distribution

Benthic macroinvertebrates not only harvest live food, but also eat dead and decaying particles of organic material, such as leaves, stems, fruits, animal carcasses and faeces. These materials are a very important source of food to many animals, and often more important than food growing in the river itself.

Plant litter and other coarse debris that fall into stream channels, fine particulates that originate from many sources including the breakdown of larger particles, and dissolved organic matter constitute the three main categories of non-living organic matter. Some of this material originates within the stream. Collectively, these sources can substantially exceed the energy transformed within streams by photosynthesis (Allan, 1995). Heterotrophic pathways are of greatest importance where the opportunities for photosynthesis are least (Vannotte *et al*., 1980).

Almost all biologically useful energy on Earth comes from plant life. Some of it is consumed directly, but most plant material dies and decays. Fungi and bacteria decompose the decaying matter, and in the process cycle essential nutrients back to a mineral form to be consumed again by algae.

Particulate and dissolved non-living organic matter are important energy inputs to most food webs and this is especially true in running water ecosystems. While primary production by the autotrophs of running waters can be substantial, much of the energy support of lotic food webs derives from non-living sources of organic matter (Allan, 1995). Heterotrophic productions require a source of non-living organic matter and the presence of micro-organisms to break the organic matter down and release its stored energy (Allan, 1995).

The division of non-living organic energy into size classes is useful in studying detrital dynamics in streams. The usual categories are Coarse Particulate Organic Matter (CPOM, greater than 1 mm), Fine Particulate Organic Matter (FPOM, less than 1 mm and more than 0.5 µm) and Dissolved Organic Matter (DOM, less than 0.5 µm) (Allan, 1995).

2.2.2.1 Coarse Particulate Organic Matter (CPOM)

Particles of organic material larger than 1 mm are considered coarse: leaves, needles, other plant parts, large aquatic plants, and the carcasses and faeces of animals. These coarse pieces are broken down into finer pieces by bacteria and invertebrates. The consumption of autumn-shed leaves in woodland streams by various invertebrates is the most extensively investigated trophic pathway involving coarse particulate organic matter (Cummins, 1973; Anderson & Sedell, 1979; Cummins & Klug, 1979; Allan, 1995).

2.2.2.2 Fine Particulate Organic Matter (FPOM)

Fine organic material (0.0005 - 1.0 mm) derives from the breakdown of coarse particulate organic matter, the faeces of animals that feed on coarse particulate organic matter, detached bits of algae or other small organic layers on the stream bottom, and forest litter and soil.

2.2.2.3 Dissolved Organic Matter (DOM)

These are the invisible particles, such as molecules of various compounds like carbohydrates, fatty and amino acids and other compounds. These compounds go into solution when water contacts soil, plant or aquatic organic matter. This is not a direct food source for benthic macroinvertebrates, but serves as food for micro-organisms and is ingested with other food sources.

2.2.2.4 Live Organic Matter.

The periphyton is an important food source to some invertebrates, particularly in shallow streams with minimal shading. In addition, organic micro-layers occurring on stones and other substrates have been shown to be a food source for aquatic insects (Rounick & Winterbourn, 1983) and to be sites of active microbial uptake of dissolved organic matter (Dahm, 1981).

Biological changes along the river continuum are many. As initially conceived for a temperate woodland stream (Vannote *et al.* 1980), low-order sites are envisioned as shaded headwater streams where inputs of coarse particulate organic matter provide a critical resource base for the consumer community. As the river broadens at mid-order sites, energy inputs are expected to change. Shading and coarse particulate organic matter inputs will be minimal, and ample sunlight should reach the stream bottom to support significant periphyton production. In addition, biological processing of coarse particulate organic matter inputs at upstream sites are expected to result in the transport of substantial amounts of fine particulate organic matter to downstream ecosystems. Macrophytes become more abundant with increasing river size, particularly in lowland rivers, where reduced gradient and finer sediment form suitable conditions for their establishment and growth. However, in general, it is true that in high-order rivers the main channel is unsuitable for macrophytes of periphyton due to turbidity, swiftness of current and scarcity of stable substrates. The only autochthonous production is by phytoplankton and they are likely to be severely limited by turbidity and mixing. Allochthonous inputs of organic matter are thus expected to be the primary energy source in large rivers. In their original formulation, Vannote *et al.* (1980) emphasized energy inputs in the form of fine particulate organic matter imported from upstream systems. Later studies have also considered also the role of lateral inputs from the floodplain (Minshall *et al*., 1984). Most obviously, shredders should prosper in low order streams and grazers in mid-order streams. Low-order streams exhibit the lowest ratio of production to respiration (P/R ratio) and highest CPOM:FPOM ratio. Proceeding downstream, a steady decline in CPOM:FPOM ratio, and a mid-order peak in P/R ratio is anticipated. Thus, heterotrophic inputs should dominate especially headwaters and large rivers, while autotrophy should play a greater role in mid-order streams. Lastly, in mid-order streams the variety of energy inputs appears to be greatest; as a consequence one might expect also to find a peak in biological diversity (Allan, 1995).

Living at either end of the continuum is not easy for aquatic creatures. The environment is harsh, with less space, less food, as well as greater extremes of temperature. It is in the middle of the continuum where there are more opportunities for making a suitable living.

It has been suggested that the relative availability of food resources changes predictably from headwaters to river mouth, causing food webs also to vary in a predictable fashion (Vannote *et al.,* 1980).

2.2.3 Functional feeding groups

Functional feeding groups are adapted to feeding on different kinds of food and so, by definition, also use different food-gathering techniques. In rivers, the basic invertebrate functional feeding groups are shredders, grazers, collectors, and predators. In the view presented by the RCC, if different parts of a river are dominated by different kinds of food, the invertebrates will occur in varying ratios of functional feeding groups down the length of the stream (Davies & Day, 1998).

2.2.3.1 Benthic macroinvertebrates in the upper reaches

Benthic macroinvertebrate communities in these areas consist mostly of a mix of collectors and shredders. The shredders consume the coarse particulate organic matter, and the collectors gather or filter the resulting fine particulate organic matter being swept downstream or settling in pools and backwaters. A few grazers harvest the mosses and the sparse shade-tolerant periphyton. Clinging and sprawling guilds dominate in the swift water. Macroinvertebrate abundance and diversity are low, reflecting low biological productivity and lack of diversity of habitats and food sources.

2.2.3.2 Benthic macroinvertebrates in the mid-reaches

The diverse food and habitats available in the mid-reaches produce abundant and diverse benthic macroinvertebrate communities. Grazers take advantage of the periphyton growth, and along with the collectors, dominate, although shredders are still common. Clingers, crawlers, swimmers and burrowers are all well represented in the diverse conditions. Macroinvertebrate abundance and diversity are high, reflecting the diverse biological productivity and habitats.

2.2.3.3 Benthic macroinvertebrates in the lower reaches

Benthic macroinvertebrate communities in the lower reaches are limited. Filtering and gathering collectors dominate, reflecting the phytoplankton and fine particulate organic matter food sources. Burrowing types dominate in the soft sediment. Macroinvertebrate abundance and diversity are low, reflecting the low biological productivity on the bottom and the lack of diversity of habitats and food sources.

The guild concept is useful because it provides a reasonable degree of subdivision in feeding roles for both invertebrate and vertebrate consumers in the streams, where the high degree of polyphagy frustrates adequate subdivision using food type alone. The particular species in a guild may change seasonally or geographically with little effect on trophic function (Allan, 1995).

2.2.3.4 Shredders

Shredders consume coarse particulate organic matter, such as leaves and other plant parts that fall into the river. Their mouthparts (especially the mandibles) are well developed for shredding and chewing. Invertebrates that feed on decaying leaves include crustaceans, molluscs and several groups of insect larvae (Cummins *et al*., 1989). The latter include Tipulidae larvae (Diptera) and larvae of several families of Trichoptera (Limnephilidae, Lepidostomatidae, Sercostomatidae, Oeconesidae) and Plecoptera (Peltoperlidae, Pteronarcidae, Nemouridae) (Allan, 1995). Tipulidae and many Limnephilidae consume, both mesophyll and venation of the leaf, whereas Peltoperlidae larvae avoid venation and concentrate mainly on mesophyll, cuticle and epidermal cells (Ward & Woods, 1986). The radula of snails and mouthparts of *Gammarus* are most effective at scraping softer tissues and the bigger crustaceans are able to tear and engulf larger leaf fragments (Anderson & Sedell, 1979).

Invertebrate detritus feeders without doubt prefer leaves that have been conditioned by microbial colonization compared to uncolonized leaves. The nutritional quality of leaves is intimately linked with the micro-organisms that contribute greatly to leaf breakdown (Allan, 1995). A higher individual growth rate is the benefit of converting ingested leaf biomass into consumer biomass (Lawson *et al*. 1984). Micro-organisms may enhance the preference and nutritional quality of leaves in at least two distinct ways (Barlocher, 1985). One, microbial production, refers to the addition of microbial tissue, substances, or excretions to the substrate; in essence the role originally proposed by Kaushik and Hynes (1971). The second potential role for micro-organisms concerns microbial catalysis, and includes all changes that render the leaf more digestible (Allan, 1995). The ability to synthesize cellulase and thereby derive nutrition from plant cell wall polysaccharides occurs in some detritus feeders, including representatives of the molluscs, crustaceans and annelids (Monk, 1976). Aquatic insects in general show negligible enzymatic activity toward cellulose and other plant structural polysaccharides, and this has been a principal reason for arguing the importance of microorganisms as an energy source (Allan, 1995).

2.2.3.5 Collectors

This functional feeding group can be divided between filtering collectors, which feed on small bits of organic matter by filtering them from the passing water, or as gathering collectors that eat small bits of organic matter from the stream bottom. Filtering collectors either have filtering hairs or fans on their bodies, or they spin some sort of silk net. The hairs, nets and fans trap food which the creatures then scour off with special upper lips or combs on their mandibles, while the gathering collectors seem to be able to use a variety of mouthparts with no special gathering adaptations.

The amount and kind of suspended particles available influences the distribution of suspension feeders. This, in turn, will depend not only on environmental features, such as lake outlets, but also on the food processing activities of other consumers (Allan, 1995).

The link between collectors, fine particulate organic matter and bacteria depends on fine particulate organic matter captured from suspension or from the substratum. Fine particulate organic matter is as yet a poorly characterized food source, and it originates in a number of ways. Categories considered to be among the richest in quality include sloughed periphyton, organic micro-layers and particles produced in the breakdown of coarse particulate organic matter. Morphological and behavioural specializations for filtering collectors are diverse and well studied (Wallace & Merrit, 1980), while gathering collectors are less well known (Berg, 1994; Wotton, 1994).
Among the macroinvertebrates in swifter streams, representatives of the Ephemeroptera, Trichoptera, Diptera, crustaceans and gastropod molluscs are prominent gathering collectors. In slower currents and finer sediments one would expect in addition oligochaetes, nematodes and other members of the meiofauna (Allan, 1995).

Trichoptera of the families Philoptamidae, Psychomyiidae, Polycentropodidae and Hydropsychidae spin silken capturing nets in a variety of elegant and intricate designs. Most are passive filter feeders, constructing nets in exposed locations, but some nets act as snares or as depositional traps where undulations by the larvae create current (Allan, 1995). Philoptomidae spin baggy silken nets to capture fine particles in large streams and rivers.

Impressive as the nets of Trichoptera are, they are but one of the many specialized adaptations for capturing particles from suspension aquatic invertebrates (Walles & Merritt, 1980). These various devices are collectively referred to as filtering adaptations, but the actual details of particle capture and retention may be more complex (Rubenstein & Koehl, 1977). Larvae of the Ephemeroptera, for example *Isonychia bicolor,* capture small particles with a fringe of hairs on the forelegs. The larvae of Simulidae are highly specialized suspension feeders. Larvae attach to the substrate in rapid, often shallow water. The paired cephalic fans of suspension-feeding larvae each consist of primary fans, which are the main suspension-feeding organs, while secondary and medial fans act to slow and deflect the passage of particles. Food items are removed by the combing action of mandibular brushes and labral bristles, further adaptations to a filtering existence, but lacking in some Simuliidae species that scrape substrates instead. Fans are open for feeding and closed at other times (Crosskey, 1990). Other Diptera families with representatives adapted to a suspension-feeding existence in running waters include the Culicidae, Dixidae and Chironomidae (Wallace & Merritt, 1980; Berg, 1994). Some Chironominae construct tubes or burrows with catch nets and create current by body movements; others such as *Rheotanytarsus* passively suspension feed by means of a sticky secretion supported by rib-like structures on the anterior end of the case (Allan, 1995).

Mechanisms of gathering collectors of fine particulate organic matter are either less diverse in comparison to the gathering collector mode, or less is known about the subject. This feeding role is well represented in most running water environments in terms of both species and total abundances. In addition to their particular food-gathering morphologies, these taxa differ in their ability to produce mucus, in mobility and body size, in their digestive capabilities, and in whether they are surface-dwellers or live within the sediments (Allan, 1995).

2.2.3.6 Grazers

Grazers remove and feed on rich carpets of algae and microbes, attached to rocks or log surfaces. Their mandibles are blade-like and adapted to scraping algae from smooth surfaces. Scraping of surface is an important feeding role; complete with specialized mandibles like Trichoptera larvae, *Neophylax, Helicopsyche* and

Glossoma, the Coleoptera larva *Psephenus* and some Ephemeroptera like *Stenonema* with brush-like mouthparts scraping algae from the stream bottom. Snails are also algal grazers.

Microbes rather than the substrate have traditionally been considered the prime source of energy. As was true for coarse leaf particulate organic matter in streams, this may be too one-sided a view, although the efficiency with which food is absorbed undoubtedly varies from microbes and small autotrophs at one extreme to lignin-rich, refractory detritus at the other. Browsing on easily assimilated organic layers may allow consumers to meet their energy needs without having to ingest large quantities of material (Lopez & Levinton, 1987). However, the issue of grazing or herbivory is more complex, partly because the resource category includes organic layers, loose fine particulate organic matter, and diatoms differing in the degree to which they stick to substrates and partly because the functional morphology of this consumer group is under-studied (Allan, 1995).

2.2.3.7 <u>Predators</u>

Animal prey represents a high quality resource for predators and parasites capable of locating and capturing other animals (Allan, 1995). Most predators engulf their prey entire, or in pieces, but some Hemiptera and Rhagionidae (Diptera) have piercing mouthparts (Cummins, 1973). Other distinctions can be made between hunting by ambush versus searching (Peckarsky, 1984), and whether prey is obtained from suspension, as in large Hydropsychidae, or strictly from the substratum, as in Platyhelminthes. Odonata and Megaloptera are seen as the dominant arthropod predators in the river system.

Occasional predation probably is widespread, particularly the ingestion of micrometazoans, protozoans and early stages of macroinvertebrates. Such unpremeditated carnivory may provide high quality protein and may also form an important link between microbial and macro-consumer food webs (Allan, 1995).

From the above descriptions, it is apparent that benthic macroinvertebrates cannot be factiously classified into functional feeding groups. Allan (1982) and Hynes (1941) put them in this perspective. Predaceous Plecoptera ingest more periphyton and detritus when small, and more animal-prey when large. An increased utilization of periphyton relative to detritus has been noted in grazing Trichoptera and Ephemeroptera as they develop. Changes in food availability obviously play a potentially large role in determining the seasonal and spatial groups. High growth rates of shredders in autumn demonstrate the close linkage of life cycle timing to seasonal changes in food availability (Allan, 1995).

2.2.4 Biotopes

In many cases the stream must be evaluated according to the different types of flow regimes present and these are call biotopes:

2.2.4.1 Riffles

Riffles are high velocity areas with turbulent flow, indicated by broken water surfaces and typically shallow water depth relative to bed particle sizes. The substratum is predominantly cobbles and boulders, with limited deposition of fine particulate matter. There are generally noticeable changes in slope from head to foot of riffles. Riffles are spatially and temporally variable in that they can “migrate” upstream or downstream with changes in flow and can become runs at high flows (Brown & Dallas 1995).

2.2.4.2 Pools

Pools possess features with slow through-flow of water that is deep relative to river size and have low to zero velocity. The substratum ranges from bedrock to sand. The flow is smooth apart from small areas of turbulence at the heads of some pools. The combination of deep water and low velocity often promotes deposition of fine particulate matter on the pool bottoms, such as sand silt and organic detritus. Pools form bodies of standing water when river flow is very low or at zero flows, and can effectively become runs at very high flows (Brown & Dallas, 1995).

2.2.4.3 Runs

Runs are features that represent areas of transition between pools and riffles/cascades. Depths are variable from fairly shallow to deep. The velocities are generally moderate, but can be low or high depending on flow conditions. The substratum conditions are variable. Runs are characterised by tranquil smooth flow with no broken surface water, no obvious changes in river bed gradient and higher ratio of depth to river-bed-roughness elements than for riffles (Brown & Dallas, 1995).

2.2.4.4 Backwaters

Areas that are hydraulically detached from the channel, as there is no through flow of water, but where water tend to enter and exit using the same route. Depth is variable, and velocity is low, often zero. The substratum is variable, but usually includes depositions of sand, silt and detritus. Backwaters tend to be situated along the margin of the main channel, often resulting from the isolation of side flood channels when flow decreases. Increased flows turn some backwaters into runs, and accumulated matter is flushed out during these periods (Brown & Dallas, 1995).

2.2.4.5 Cascades

Features characterised by free-falling water over bedrock and boulders in step-like arrangements. Water depths and velocity are not distinguishing features. They usually consist of a series of low waterfalls with downstream pools. The average gradient is steep and the elevation of the substratum is a distinguishing criterion with step height a maximum of 3 m (Brown & Dallas, 1995).

2.3 BIOINDICATORS OF WATER QUALITY

2.3.1. Definition

Biomonitoring is the monitoring of living organism, usually as indicators of habitat integrity, while bioassessment is the use of living organisms to assess these conditions (Davies & Day 1998). Biomonitoring can be traced back in history to Aristotle, who placed freshwater fish into seawater to observe their reactions (Rosenberg, 1998). The first water toxicity experiments were published in 1816, and described longer survival of several species of freshwater molluscs in 2% than in 4% saline solutions (Rosenberg 1998). The history of bioindicator systems for surface water quality assessment started in the 1850 by Kolenati and Cohn who observed that organisms occurring in polluted water were different from those in clean water (Iliopoulou-Georgudaki *et al*., 2003). Furthermore, the use of community structure of freshwater organisms for bio-monitoring can be traced back to the pioneering work of two German scientists, R. Kolkwitz and M. Marsson, in the early 1900’s. Their publication on saprobity (degree of pollution) led to the development of indicator organisms (Rosenberg, 1998). Today, indicators are utilised to summarize a wide variety of states, from biological health to economics of water management.

Rapid assessment approaches differ from the traditional statistically based studies, in that they are usually characterized by involving more than one type of measurement, and some summarization of these measurements is used to compare them with pre-determined thresholds, rather than relying on statistical comparisons of the individual measures. This technique is referred to as the multi-metric approach. The multi-metric approach involves defining an array of measures, or metrics, that each provide information on diverse biological attributes and when integrated, provide an overall indication of the condition of the biological community (Barbour *et al*., 1995).

The concept of bioindicators can be defined as a species or assemblage of species particularly well matched to specific features of the landscape and/or reacting to impacts and changes (Paoletti & Bressan, 1996). Instead of focusing on a few indicator species, more reliable information can be obtained from studies of a set of species or one or more higher taxon, with measurements made not at the level of presence/absence but as numbers, biomass and dominance. The use of guilds such as detritivores, predators, pollinators, parasitoids, decomposers, scavengers, collectors, gatherers as bioindicators can reveal interesting differences in the assessment of a natural habitat (Holopainen & Oksanen, 1995). Even the disappearance of a single species from a landscape can give extensive information such as a complex combination of events occurring, including the collapse of meta-population or events such as field dimension, tillage, field contamination, pollution, impoundments, chemical sprays or spillage (Burel, 1995). Over time, the use of these bioindicators has become an important concept in the process of assessing damaged and contaminated areas as well as industrial and urban settlements (Paoletti 1999) and lately is being used more and more in the aquatic environment.

Insects as a group are most suitable for the assessment due to the fact that they occur in almost every imaginable habitat, and are important in the ecological functioning of natural ecosystems through diverse activities, ranging for decomposition of organic matter to provision of food for higher taxa. In fact, insects play roles as predators, parasites, herbivores, saprophages, and pollinators, among others. This indicates the pervasive ecological and economic importance of these taxa in both aquatic and terrestrial ecosystems (Rosenberg *et al*., 1986). It is because of these compelling reasons that freshwater macroinvertebrates enjoy popularity in bio-monitoring practice as they offer a number of advantages (Rosenberg 1998) in that**:**

1. They are ubiquitous, so they are affected by perturbations in many different habitats;
2. They are species rich, so the large number of species produces a range of responses;
3. They are sedentary, so they stay put, which allows determination of the spatial extent of a perturbation;
4. That they are long-lived, which allows temporal changes in abundance and age structure to be followed; and
5. That they integrate conditions temporally, so like any biotic group, they provide evidence of conditions over long periods of time.

Nonetheless, the sheer number of insects, leads to major challenges in completely utilising this goal and insects are often avoided for a variety of reasons. As insects are seen as small and cryptic in their colouration and behaviour, during an environmental impact assessment they do not receive the attention given to more conspicuous animals such as fish, birds, or mammals, as well as difficulty in keeping invertebrates in captivity for laboratory bioassays or toxicity/tolerance tests (Chutter & Health, 1993; Rosenberg & Resh, 1993). A notion also exists that the costs of using insects outweigh the benefits (Resh & Gradhaus, 1983). Certainly, the analysis of collections of insects is often labour-intensive and time-consuming, but new approaches are developed to deal with this problem (Rosenberg *et al*., 1986). A further problem is that development of taxonomic keys at species-level identification is needed for work in environmental impact assessments (Resh & Gradhaus, 1983) and basic guidelines for the use of insects in assessing environmental disturbance is generally lacking.

The use of wild stock in tests is also hampered by the usual lack of historical information, such as genetic variation (within species), health status, previous exposure and age differences (Snell, 1990).

2.3.2 World perspective’s

European countries are seen as the leaders in using macroinvertebrate community assessment as a planning tool for managing water uses, for ambient monitoring, and for evaluating the effectiveness of pollution control measures (Metcalfe, 1989). In the United Kingdom, the models predominantly make use of multivariate statistics in which a few environmental variables considered to be unaffected by human activities are used, and from which predictions are made for the fauna expected at a given test site (Resh *et al*., 1995).

In France, use is made of the Ephemeroptera, Plecoptera, Trichoptera and Coleoptera (EPTC) species richness and the Indice Biologique Global Normalisé (IBGN) (Compin & Cereghino, 2003).

American researchers analyse data using several indices presumed to represent ecological features of interest (Resh *et al*., 1995). This includes the interpretation of the potential confounding effects of habitat degradation and water quality (Plafkin *et al*., 1989) and to a lesser extent the prediction of biotic communities expected at a given site (Winget & Mangum, 1979). The monitoring programmes that have been developed in the United States include the Environmental Monitoring and Assessment Program (EMAP) of the Environmental Protection Agency (1990), the National water-Quality Assessment Program (NAWQA) of the Geological Survey (Gurtz, 1994), the Biomonitoring of Environmental Status and Trends (BEST) Program of the United States Fish and Wildlife Service, and an inter-agency oversight committee called the Intergovernmental Task Force on Monitoring Water Quality (Anonymous, 1992). Other influences on the use of benthos in bio-monitoring include the National Biological Survey of the United States Department of Interior (Anonymous, 1993).

2.4 SOUTH AFRICAN SCORING SYSTEM

In South Africa, in-stream invertebrate bioassessments are undertaken on the basis of a modification of the methodology used by the British Monitoring Working Party (BMWP). This system is referred to as the South African Scoring System (SASS). This comprises a simplified field manual intended to reduce both the time and expertise required for identification of organisms and is currently in its fifth version (Chutter, 1994; Thirion *et al.,* 1995; Chatter, 1998; Dickens & Graham, 2002).

The RIVPACS, the utilised method of the British Monitoring Working Party in the United Kingdom, developed by Wright *et al*. (1984) is seen as the model for suitable methodology in river assessments. One of the best-known examples of countries that adapted the system to great success is Australia (Dickens & Graham, 2002). South Africa has also experienced a surge of support for this type of river assessment, expanding from rather hesitant beginnings nearly three decades ago when Chutter (1972) developed a Biotic Index. This index was never widely used, as it was excessively labour intensive (Chutter 1998; Dickens & Graham, 2002). In the 1990's Chutter set out to develop an index that would be faster and easier to use, basing it on the BMWP method. His index, called SASS (South African Scoring System), evolved through several phazes of refinement, received the input of a large group of practitioners via a SASS Forum hosted by the South African Water Research Commission (Chutter, 1994; 1998; Dickens & Graham, 2002). This system allows a rapid bioassessment, which is not labour intense and is less expensive than the chemical analysis of water samples and indicate water quality variation over a period of time (Chutter, 1995).

SASS is suitable for the assessment of river water quality and river health. Roux (1994) stated that this biomonitoring method can also be used to:

- Assess the ecological state of aquatic ecosystems;
- Assess the spatial and temporal trends in ecological state;
- Assess emerging problems;
- Set objectives for rivers;
- Assess the impact of developments;
- Predict changes in the ecosystems due to developments; and
- Contribute to the determination of the Ecological Reserve.

The method is designed for low or moderate flow hydrology and is not applicable in wetlands, impoundments, estuaries and other lentic habitats. It has also not been sufficiently tested in ephemeral rivers and should thus be used with caution. The method works best when the diversity of biotopes is wide and includes riffles or rapids, but it also produces valuable results from poor habitats. It is necessary to interpret the data in relation to habitat quality, availability and diversity and ultimately also in relation to the eco-region from which the study area come. It is also necessary to interpret the data in relation to the season of collection, as some natural variation will occur during the course of the year and between years (Dickens & Graham, 2002).

SASS5 forms the backbone of the National River Health Programme (Uys *et al*., 1997) and is increasingly being included in the determination of the Ecological Reserve as required by the South African National Water Act. SASS5 has been recommended for the determination of the flow requirements of rivers (O'Keeffe & Dickens, 2000) and has also been used for many impact assessments, such as reported in Dickens & Graham (1998). Other institutions such as Cape Metro Council, Umgeni Water, Umlaas Irrigation Board, Mpumalanga Parks Board, the Department of Water Affairs and Forestry, CSIR and many others, including many forestry companies and heavy industries, currently make use of this bioassessment method for determining their water quality (Dickens & Graham, 2002).

2.4.1 **Other sampling methods reported in the literature:**

- Trent Biotic Index (Woodiwis, 1978);
- Extended Trent Biotic Index (Woodiwis, 1978; Metcalfe, 1989);
- Extended Biotic Index (EBI) (Woodiwiss, 1978);
- UK Biological Monitoring Working Party (BMWP) score (Chesters, 1980; Vigano *et al*., 2002);
- Belgian Biotic Index (De Pauw & Vanhooren, 1983);
- UK Average Score Per Taxon (ASPT) (Armitage *et al*., 1983; Vigano *et al.*, 2002);
- Modified Extended Biotic Index for Italian rivers (Dickens & Graham, 2002);
- Iberian ASPT & BMWP (Rodriguez & Wright, 1987);
- Lincoln Quality Index (LQI) (Extence *et al*., 1987);
- Modified Family Biotic Index (FBI) (Plafkin *et al*., 1989; Mackie, 2004);
- Environmental Monitoring and Assessment Program (EMAP) of the US EPA (Anonymous, 1990);

- Biomonitoring of Environmental Status and Trends (BEST) (Anonymous, 1992);
- National water-Quality Assessment Program (NAWQA) (Gurtz, 1994),
- Indice Biotico Esteso (IBE) (Dickens & Graham, 2002);
- A method of habitat assessment developed by McMillan (1998);
- Ephemeroptera, Plecoptera, Trichoptera and Coleoptera (EPTC) species richness (Compin & Cereghino, 2003); and
- Indice Biologique Global Normalisé (IBGN) (Compin & Cereghino, 2003).

2.5 MACROINVERTEBRATES

Out of the estimated 1.4-1.8 million species that have been identified (Hammond, 1995), macroinvertebrates are the majority of organisms. An estimation of actual living arthropod species ranging from 12.5 million to over 100 million, makes them particularly suited for the use in environmental impact assessment due to their high species diversity, ubiquitous occurrence, and importance in the functioning of natural ecosystems (Rosenberg *et al.*, 1986).

Freshwater Arthropoda are contained in three subphyla, Uniramia, Chelicerata and Crustacea, and are all diverse and important components of river ecosystems. Arthropods occupy every heterotrophic niche in the benthic habitat of most permanent and temporary aquatic systems. These metameric coelomates are characterized by a chitinous exoskeleton and stiff, jointed appendages modified as legs, mouthparts, and antennae (except in water mites) (Thorp & Covich, 1991).

The insects exhibit the greatest diversity in form and habit, occupying every kind of freshwater habitat imaginable, including temporary streams and ponds, the shallowest and deepest areas of lakes, the most pristine and polluted rivers, roadside ditches, eaves troughs, moss, within and on macrophytes and exposed to all ranges of water chemistry, from acidified to alkaline bodies of water (Mackie, 1998). Aquatic insects are one of the most important components of aquatic ecosystems. They provide food for many vertebrates as well as feeding upon large numbers of lesser animals (McCafferty, 1981). Only a few insects, 3-5%, have adapted successfully to the aquatic environment (Daly *et al.*, 1997).

They also represent all the functional feeding groups, including predators, shredders, grazers, (or scrapers), filter feeders, gatherers, piercers and parasites (Mackie, 1998). Furthermore these macroinvertebrates are rather habitat specific (Zwick, 1992).

In conjunction with their habitat specifications, most macroinvertebrates are very sensitive to environment changes and it is on this basis that Chutter (1994) developed the SASS bioassessment. In this assessment, Chutter used mostly the presence of families of the insect orders, Ephemeroptera, Trichoptera, Coleoptera, Hemiptera, Odonata, Diptera, Plecoptera, Lepidoptera and Megaloptera. He also included the presence of the taxa Hydracarina, Turbellaria, Amphipoda, Decapoda, Oligochaeta, Hirudinae, Gastropoda and Bivalvia, each with values to their respective tolerance levels to pollution.

2.5.1 Ephemeroptera

Ephemeroptera (Mayflies) are found throughout the world except Antarctica (Edmunds & Waltz, 1996) and consist of over 2 000 named species in 200 genera and 19 families. They date from the Carboniferous and Permian times and are regarded as the oldest of the existing winged insects. They are unique in being the only insect order with a sub-imago, which is usually short-lived and sexually immature with wings (Brittain, 1982). The order derives its name from the Greek word *ephemeros* meaning "lasting but a day," due to the imago having a very short lifespan (Harker 1989).

The first stage of the life of an Ephemeroptera is the larvae, which looks significantly different from the imago, with much longer antennae, functional mouthparts as well as being aquatic (Brittain, 1982). When the larvae hatch from the eggs, they are less than 1 mm long. At first they have no gills and their body shape varies according to habitat. Ephemeridae have large frontal tusks and burrow into the soft substrate of lakes and rivers. In rivers and streams some species have streamlined bodies enabling them to swim through the current. Others, members of the Heptageniidae, are dorso-ventrally flattened and attach to rocks to avoid being swept away by the current (Williams & Feltmate 1992). This has reached the ultimate level in the genus *Rithrogena* with gills modified to act as a suction cup to hold the larvae on the rocks (Edmunds & Waltz 1996).

As they develop, Ephemeroptera larvae go through a large number of moults, with most species having 15-25 instars. Estimates for some species reach as many as 50 and variations occur within single species (Williams & Feltmate, 1992), giving a larval life cycle that can range from 3 weeks to 2.5 years depending on the species (Brittain 1982). The number of moults a larvae undergoes does not depend on its nutrition, but on the increase in size that comes with each moult (Harker, 1989).

Older Ephemeroptera larvae are characterised by an elongated body, large head, well-developed mouthparts and stout legs, with three distinct cerci, occasionally two (McDonald *et al.*, 1990) and may grow to anything from 4 mm to 3 cm long (Harker, 1989). Paired gills on the abdomen are the most characteristic feature distinguishing Ephemeroptera larvae from other aquatic insects (Gerber & Gabriel, 2002). These gills beat to control the flow of water through the body so as to control the amount of oxygen and salt flowing over the body. Larvae in still waters generally have larger gills, than those in running water. This allows the larvae of each habitat to get an optimum flow of water (Harker, 1989).

Most Ephemeroptera larvae are herbivores, feeding on algae and diatoms, or detritivores feeding on detritus. Some species are collectors, filter feeding on floating material, while others are scrapers actively removing plant material from the rocks. Some omnivorous species exist in the genera *Isonychia*, *Siphlonurus* and *Stenonema*. There are even some carnivores in the genera *Dolania*, *Spinadis* and *Raptobaetopus* of which the first two at least feed on Chironimidae larvae (Brittain, 1982).

Some signs of reproductive development can be seen in the last few stages of the larvae, even long before it becomes an adult. At this stage, male larvae have the beginnings of clasping organs on the lower portions of their abdomen, with which they hold the female during copulation. In some species, the males have divided eyes. The upper portion is for seeing movement, and the lower portion is specialized for seeing details. The female has smaller eyes and the oviduct is situated in the lower abdomen (Harker, 1989).

During the last moult the guts are voided and the mid-gut section fills with air. Often, many larvae will then simultaneously let go of their hold on their anchor in the water and float up to the top. Once they reach the surface, the cuticle ruptures on the thorax and the wings appear. The sub-imago is a brief transitional stage that moults again into a sexually mature adult (Brittain, 1982; Harker, 1989).

The sub-imago has microtrichia on the wings and on the body; the wings are dull and pigmented. After a short rest, it flies from the water to some form of shelter and moults again within 24 to 48 hours. This additional stage enables the legs and cerci of the insect to grow longer. Longer cerci provide more stability in flight, and longer legs make it easier for the male to grasp the female in mating (Harker, 1989).

If conditions are favourable for large numbers of larvae to synchronously reach maturity, an impressive mass emergence of hundreds of thousands of sub-imagoes and swarms of adults may occur (Edmunds & Waltz, 1996). Emergence may also be rhythmical i.e. *Povilla adusta* which exhibits lunar rhythm to its emergence in several African lakes (Edmunds, 1988). Males and females usually tend to emerge at the same time (Edmunds, 1988).

The imago resembles the larvae, but possesses very short antennae, vestigial mouthparts, usually only two cerci and two pair of hairless wings held vertically in rest, with the second pair being considerately smaller than the first and absent in some species altogether (Brittain, 1982; McDonald *et al.*, 1990). They also lack functional gut and in one species, *Lachlania saskatchewanensis*, females lack functional legs (Edmunds & Waltz, 1996).

The imago typically survives for only a few hours with the males flying in aerial undulating swarms 5-15 meters above ground. Females fly into the swarm and are caught by a male. Copulation takes place in flight, and the female usually lays her clutch of eggs within minutes or hours. Males die shortly after mating; females usually die soon after oviposition (Brittain, 1982).

Females lay their eggs by flying low over the water surface and dipping an extruded egg sac into the water or dropping eggs on the water surface where the eggs sink to the bottom (McDonald *et al.*, 1990). In some species inhabiting swift flowing water, the female will crawl under the water surface to lay her eggs directly on submerged rocks. The duration of the egg stage can range from a few days to some months (Edmunds & Waltz, 1996). In most species the female lays between 500 and 3000 eggs but some species of *Palingenia* can produce as many as 12000 (Brittain, 1982). The greatest diversity of Ephemeroptera is found in low temperate rivers and streams. Generally Ephemeroptera require unpolluted, well oxygenated, cool water to survive (Edmunds & Waltz, 1996). Ephemeroptera larvae are characteristic of shallow streams and littoral areas of lakes, and are widely distributed. However, many species are restricted to specific substrata of macrophytes, moving stream areas, or sediments of specific sized particles (Williams & Feltmate, 1992).

Ephemeroptera have attracted human attention for a long time and Swammerdam did one of the oldest studies of Ephemeroptera biology in 1675. Unfortunately the genus of Ephemeroptera studied by Swammerdam, *Palingenia sp.,* is now extinct in the Netherlands and the rest of Western Europe (Brittain, 1982), proving that the Ephemeroptera has a general low tolerance to pollution.

The latter makes Ephemeroptera communities especially useful indicators of ecosystem health (Edmunds & Waltz, 1996). In many habitats, they are an important food source for fish, and are used by fly fishermen as bait.

2.5.2 Odonata

Odonata is an order of aquatic palaeopterous insects. There are about 6500 existing species in just over 600 genera (Watson & O'Farrell, 1991) classified in 25 known families (Corbet, 1999). Odonata can be divided into three sub-orders: Anisoptera (dragonflies), Zygoptera (damselflies) and Anisozygoptera, a relic group represented by only two living species found in Japan and the Himalayas (Watson & O'Farrell, 1991). The major two suborders occur in every faunal region in the world, except for Antarctica (Williams & Feltmate, 1992).

The larval stage is mainly aquatic, and shows very little similarity to the imago stage. They are strictly predaceous and occur in all types of inland waters (Watson & O'Farrell, 1991). They can be divided into three ecological guilds: burrowers, sprawlers, and climbers (Williams & Feltmate, 1992).

The most obvious characteristic shared by all Odonata larvae is a conspicuous grasping labium, used for capturing prey (Corbet, 1980) and with posterior tracheal gills (Corbet, 1999) that can be seen as three leaf-like filamentous gills at the tip of the abdomen in Zygoptera, whereas Anisoptera possess reticular folds in the anus functioning as gills. Furthermore, the larval Zygoptera abdomen is longer and narrower, while Anisoptera larvae are shorter and bulkier (Corbet, 1999).

The predacious Odonata larvae use their modified extensible labium or "mask" to capture insect, crustacean, mollusca, or oligocheata prey. They have even been reported to consume small vertebrates (Williams & Feltmate, 1992). Some species burrow in the substrate and ambush prey that they detect by tactile or vibrational signals. Other species actively stalk their prey. Odonata is one of the few orders of aquatic insects whose immature stages have eyes well developed for hunting (Williams & Feltmate, 1992).

Growth in the larval stages is quite variable, depending on altitude and latitude, especially in relation to temperature and food supply. Larval development varies within a range of about five weeks to five years, with 10 to about 20 instars (Williams & Feltmate, 1992). The fully developed larva crawls up out of the water and moults one last time, emerging from its exuvium as an imago with functional wings (Corbet, 1999).

In any given set of climatic conditions most species emerge over a period of about a month, at well synchronized times of year. Many species emerge at night to avoid predation during this vulnerable time, but in colder environments some species wait for sunrise before emerging (Corbet, 1980). After emerging most Odonata leave the vicinity of the water and undergo a period of maturation, this generally lasting about one month when the gonads finish developing, and the body colour brightens (Corbet 1980). This is also the time for dispersal.

The adult stages of Odonata are medium to large in size, often conspicuous and/or brightly coloured insects. They are generally found at or near fresh water although some species roam widely and may be found far from their breeding sites (Watson & O'Farrell, 1991).

Adult Odonata are visually oriented hunters with exceptional aerobatic ability and extremely acute eyesight (Watson & O'Farrell, 1991) and generally feed on flying insects, which they catch on the wing, either by flying around constantly, or by sitting perched on a lookout post (Corbet, 1980).

Members of both major suborders have large heads with very large compound eyes relative to the rest of their body. The head also carries three ocelli, a pair of short, bristle-like antennae and mouthparts adapted for biting. All Odonata have a prehensile labium in the larval stages, which can be extended forward from underneath the head faster than most prey can react, making their bite fatal to prey (Corbet 1999). The prothorax is small, in comparison to the larger mesothorax and metathorax, fused into a single, strong pterothorax and directed in an angle, casing the weak legs, suited for perching and to holding prey but not to walking, into a capturing formation (Watson & O'Farrell, 1991). The pterothorax also carries the two pairs of transparent membranous wings with many small veins (Corbet 1999).

Identification within the Odonata is based on the wings and specifically on the venation (Watson & O'Farrell, 1991). In Zygoptera the front and hind wings are similar in shape, and as a result they fly slower than Anisoptera. Also, Anisoptera do not have hinges enabling them to fold their wings together when resting, though Zygoptera do (Corbet, 1999). In most families, a conspicuous pterostigma is carried near the wing tip (Watson & O'Farrell, 1991).

The flexible abdomen is long, rarely any shorter than the length of one wing, with 10 visible segments, and terminates in clasping organs in both sexes. Females of all Zygoptera and several Anisoptera families carry a prominent ovipositor on abdominal segments 9-10. Males always possess secondary genitalia on the underside of abdominal segments 2-3 (Watson & O'Farrell, 1991) as well as his primary sexual apparatus on the 9th segment at the tip of the abdomen.

Before mating, the male Odonata must charge the secondary copulatory apparatus with sperm from the primary copulatory apparatus. Mating commences with the male grasping the female with his abdominal claspers. The pair then assume the position with the tip of the female's abdomen, and thus her sexual apparatus, engaging the males secondary copulatory apparatus. The male first uses his penis to remove any sperm left by a previous male before inseminating her himself (Corbet, 1980).

Anisoptera copulate while in flight, the male lifting the female in the air. Zygoptera copulate while perched, sometimes flying to a new perch. The length of time required for copulation varies greatly. Aerial copulations may last mere seconds to one or two minutes. Perched copulations usually last from five to ten minutes.

Intraspecific competition amongst males for females is fierce. In some species of Odonata, the males will remove all the sperm of rival males from a female's body before transferring his own sperm. These species are equipped with a scoop at the tip of the male's abdomen used for this purpose (Corbet, 1999). Males are also extremely territorial, sometimes patrolling their territory for hours at a time (Corbet, 1999).

After fertilization, the eggs are deposited either endophytically (inside the living tissue of a plant) or exophytically (into or onto the water or the mud of the bank) (Watson & O'Farrell, 1991; Williams & Feltmate, 1992). Anisoptera eggs are round and about 0.5 mm long, whereas Zygoptera eggs are cylindrical and longer, about 1 mm long (Corbet, 1999). Eggs are normally laid in batches. Endophytically laying species tend to be limited to several hundred or less eggs per day, whereas exophytically laying species can lay several thousands at a time. Some temperate Odonata overwinter in the egg stage and thus the eggs do not hatch for several months, but in tropical species the eggs can hatch in as little as five days (Corbet, 1980).

The distribution of various groups and species of Odonata is highly variable. Some genera and species are widespread, while others are highly local in their distribution. Some families are restricted to cool streams or rivers, others to ponds or still clear waters, and some to marshy places (Corbet, 1999). The largest numbers of species are found at sites that offer a wide variety of microhabitats (Corbet, 1999).

In some countries, notably Japan, Odonata have long been a popular subject of art and culture, and rank with butterflies and birds as a topic of popular scientific interest. In the European folk tradition, Odonata are generally accorded a less favorable status as "horse-stingers" or "devil's darning needles". In fact they neither sting nor bite and all species are completely harmless. If anything, Odonata are beneficial to humans because as voracious aquatic predators they assist in the control of insect pests (Watson & O'Farrell, 1991). Fishermen use them as bait and adult Anisoptera forms a minor food item in some countries. However, Odonata is of little economic importance. Their main attraction for humans is aesthetic (Watson & O'Farrell, 1991).

The presence of Odonata may be taken to some extent as an indicator of ecosystem quality (Corbet, 1999). Moore (1997) noted their considerable potential as indicators, and that their conspicuousness renders them valuable for rapid assessment of water quality, so that a count of Odonata would provide a quick, and therefore low-cost, indication of the health or richness of the lake or river. Local faunal composition may be strongly affected by any change in water flow, turbidity, or in aquatic or waterside vegetation (Watson & O'Farrell, 1991). It is also known that Anisoptera tend to be much more sensitive to pollution than are Zygoptera (Corbet, 1999).

2.5.2 Plecoptera

Plecoptera (Stoneflies) with a worldwide distribution (Stewart & Harper, 1996), are considered to be the most primitive order of living Neoptera. Plecoptera is a small order of exopterygote insects (Harper & Stewart, 1984) that consist of about 3000 species (Stewart & Stark, 1993) in 239 genera belonging to 15 families (Williams & Feltmate 1992). Most stoneflies are classified in the guilds of clingers or sprawlers, as they are closely associated with the substrate or leaf litter (Williams & Feltmate, 1992).

All larvae are aquatic, and resemble the adults in many respects. The larvae always have long cerci and never a third median caudal filament as well as three-segmented tarsi. Gills, if present, can occur on various parts of the thorax and abdomen and are composed only of filaments, not plates (Harper & Stewart, 1984).

Plecoptera larvae are restricted to cool, clean streams with a high dissolved oxygen content. Some species, however, may be found along the wave-swept shores of large oligotrophic lakes (Williams & Feltmate, 1992).

Generally, Plecoptera species feed on algae and other submerged vegetation. Some families (Perlidae & Chloroperlidae), however, are predators of Ephemeroptera larvae and other small aquatic insects (Stewart & Stark, 1993). They are most likely to being herbivorous or detritivorous in early instars, while late instars turning to predators. These species are usually engulfers, that swallow their prey whole or bite off and swallow parts of prey. They are active searchers or pursuit predators, using their long filamentous antennae to locate prey (Williams & Feltmate, 1992).

When subjected to low dissolved oxygen concentration, the larvae of many species exhibit a characteristic push-up behaviour that increases the rate of movement past the gills. The gills are variously placed among species on the neck, thorax and abdomen. However, some species have no gills and respiration in these cases is assumed to be cuticular (Williams & Feltmate, 1992).

The larvae require anything from a year to three years, depending on the species and habitat, to complete up to 30 larval moults (Stewart & Stark. 1993) towards maturity (Stewart & Harper, 1996).

Temperate species that overwinter as larvae often keep on growing even in water temperatures close to 0°C. It seems that it is warm water temperatures rather than cold ones that punctuate Plecoptera life cycles. It is this ability to spend the summer in diapause that enables some species to live in temporary streams (Williams & Feltmate, 1992).

When the larvae are mature and ready to emerge, the final instar larvae tend to migrate towards the bank where they crawl out of the water to shed their skins (Williams & Feltmate, 1992; Stewart & Harper, 1996).

Adult Plecoptera are generally found on the banks of streams and rivers from which they have emerged. They are not active fliers and usually remain near the ground where they feed on algae or lichens. In many species, the adults are short-lived, from one to four weeks (Williams & Feltmate, 1992), and are without functional mouthparts (Stewart & Stark. 1993).

Morphologically the terrestrial Plecoptera imago consists of a somewhat flattened soft-body with long filiform antennae, at least half the length of the body, and a pair of cerci (Stewart & Stark, 1993). Providing a range in size from 5 mm to over 5 cm (Stewart & Harper, 1996). They have three segmented tarsi (Harper & Stewart, 1984). Furthermore, they have tufts of gills at the base of each leg and sometimes on the first two or three abdominal segments (Stewart & Harper, 1996).

Most imagines are winged with membranous wings, with the hind pair usually larger than frontal pair, and folded flat over the body at rest (Stewart & Stark, 1993). A few species show apterous or brachypterous characteristics in wing development (Harper & Stewart, 1984). Overall Plecoptera do not fly very well and this has prevented them from crossing even small geographical barriers (Williams & Feltmate, 1992).

During the mating season, the sexes attract each other by drumming the tips of their abdomens on the substrate. Mating occurs while the partners are clinging to vegetation or on to the ground (Stewart & Harper, 1996).

Females lay their eggs directly on the water surface or as they fly over the water. In most species the eggs hatch in three to four weeks (Stewart & Harper, 1996; Williams & Feltmate, 1992).

Field surveys clearly show that the larvae of many species are associated with particular sections of a streambed or lakeshore (Williams & Feltmate 1992). The specific microhabitat occupied depends on a variety of environmental factors such as the nature of the substratum (particle size & configuration), current regime, presence of other organisms, and local variations in water chemistry and temperature. Habitat preference often changes as the larvae develop and also with seasonal changes (Williams & Feltmate, 1992).

In general, the preferred habitat is rocky streams with a noticeable current, but there are species that live in sandy places. Cold lakes and ponds are also suitable habitats in the north and at high altitudes (Harper & Stewart, 1984). Many studies have shown that the various species are restricted to particular situations. For instance, large

Perlidae and Perlodidae are usually found on and under large stones. Chloroperlidae tend to occur in gravel while Pteronarcyidae and Nemouridae are most frequently encountered in leaf packs (Harper & Stewart, 1984).

The high water quality requirements of the Plecoptera larvae and their reaction attributes to habitats changes from low oxygen levels, siltation, high temperatures and organic enrichment, led to their effective use as biological indicators of environmental degradation (Williams & Feltmate, 1992). Furthermore, they are susceptible to human abuse of water courses. Any effluent that reduces the oxygen content of the water quickly extirpates them (Harper & Stewart, 1984). Even quite, minor pollution sources such as farm drainage can eliminate stoneflies from nearby streams. Land development or impoundment, both of which causes an increase in the summer temperature of water, can eliminate Plecoptera from the habitat (Harper & Stewart, 1984).

Plecoptera are furthermore an important source of food for game fish (trout & bass) in cold mountain streams (Stewart & Stark, 1993) and have been used extensively in historical biogeography, due to their lack in distribution ability (Williams & Feltmate, 1992).

2.5.4 Trichoptera

Trichoptera (caddisflies) are the most diverse insect order whose members are exclusively aquatic and can be found in lakes, rivers, and streams all over the world (Resh & Rosenberg, 1984). The Latin name Trichoptera comes from the Greek Trichos and Pteron, meaning hairy winged (Hicken, 1967).

Trichoptera are holometabolous insects (Williams & Feltmate, 1992) with 12 000 species, placed in 45 families and about 600 genera from all faunal regions, but it has been estimated that the world fauna may contain as many as 50 000 species (Morse, 1984).

Trichoptera are closely related to the order Lepidoptera and together the two orders comprise the super-order, Amphiesmenoptera (Kristensen, 1984; Wheeler *et al*. 2001). The order of Trichoptera can be divided into three superfamilies, which include five groups based on case-building behaviour. The three superfamilies are largely characterized by differences in the way silk is used (Ross, 1944) to produce nets or tubes, or as glue to make various types of portable cases, often incorporating sand and small pebbles, or bits of leaves and twigs, each genus or even species building its own particular style of case. Some larvae are free-living and predaceous, but nevertheless lay down a strand of silk as they move (Ross 1944). Hydropsychoidea (Hydropsychidae, Philopotamidae, Polycentropodidae, and Psychomyiidae) are net spinners and retreat makers; Rhyacophiloidea include the free-living forms (Rhyacophilidae), saddle-case makers (Glossosomatidae), and purse-case makers (Hydroptilidae), and Limnephiloidea are the tube-case makers (Williams & Feltmate 1992).

Trichoptera are generally classified in one of the following guilds: clingers, sprawlers, climbers, or burrowers (Williams & Feltmate, 1992). Trichoptera larvae, with very few exceptions, are aquatic and primarily detritivores (Mackay & Wiggins, 1979). The larvae are mostly eruciform or campodioform and have a strongly sclerotized head with very short antennae and biting mouthparts (Hicken, 1967). The larval thorax is well developed, with at least the pronotum covered dorsally by a pair of sclerotized plates (Williams & Feltmate, 1992). Furthermore, their legs are well developed with a single tarsus. The abdomen usually consists of ten segments. In case bearing species the first segment bears three papillae, one dorsally and the other two laterally, which assist in holding the insect centrally in its case, allowing a good flow of water over the cuticle and gills (Hicken, 1967). Abdominal prolegs are absent except for a pair of anal prolegs on the last abdominal segment, with each proleg being equipped with an apical anal claw (Resh & Rosenberg 1984).

Trichoptera larvae have biting mouthparts and most of them such as *Anabolia nervosa* and *Sericostoma personatum* are omnivorous, some are facultative herbivores e.g. *Limnephilus rhombicus* and *Silo nigricornis*, while a few are facultative carnivores, such as *Rhyacophila dorsalis* which is a general predator even feeding on other Trichoptera larvae (Mackay & Wiggins, 1979). Some carnivorous larvae have been known to attack animals much larger than themselves, and furthermore feed on the bodies of dead vertebrates (Mackay & Wiggins, 1979).

In most cases, the predatory species are free-living or spin silken structures in the water in the form of webs or tunnels, which they use to entrap prey. The scavengers and herbivores live within protective cases which they build from their own silk and stones, twigs, leaf fragments, or other natural materials (Mackay & Wiggins, 1979). Case design and construction is distinctive for each family or genus of caddisfly.

The larvae undergo approximately five instars before pupation and have to build a new case each time it moults (Mackay & Wiggins, 1979). Most species are univoltine, but some complete more than one generation per year, whereas others require two years for development (Williams & Feltmate, 1992).

The pupal phase last generally two to three weeks, although some species may overwinter in this form (Williams & Feltmate, 1992). Case-bearing Trichoptera pupate inside their cases, using the cases as a puparium or a cocoon by blocking of both ends (Hicken, 1967) after it has been secured to the substrate (Williams & Feltmate, 1992). Non case-bearing species secrete themselves in a cocoon of silk and bits of detritus.

Gills similar to the larval gills are normally present in all those families possessing eruciform larvae, as well as in the Polycentropidae and the Hydropsychidae (Mackay & Wiggins, 1979). Trichoptera have exarate pupae that possess dectitious mandibles (Resh and Rosenberg 1984), with which they actively cut their way out of the old larval skin and or the cocoon before rising to the surface and climbing up on to foliage or rocks above water level to prepare for emergence. The exceptions to this are some of the Phyrganeidae that do not close the anterior end of their case when pupating and do not have to chew their way out and thus posses greatly reduced mouthparts (Mackay & Wiggins, 1979).

Adult Trichoptera, in contrast to the larvae, are terrestrial and often occur in large numbers in lakeside or streamside habitats. They are mostly nocturnal, weak-flying insects often attracted to lights. During the day, they hide in cool, moist environments such as the vegetation along riverbanks.

They are described as medium-sized insects with setaceous and often long antennae (Hicken, 1967). Their mouthparts are reduced, with mandibles essentially absent, but have a well-developed haustellum formed from a fusion of the hypopharynx and labium, and used in some species to absorb liquids. The compound eyes are well developed, and ocelli may be present or absent (Resh & Rosenberg, 1984). They furthermore posses membranous wings covered with setae or hairs, and occasionally with patches of scales, when at rest wings are held roof-like over the body (Hicken 1967). The forewings are somewhat longer than the hindwings, although the hind wings may be broader (Resh & Rosenberg, 1984). The legs contain conspicuous tibial spurs (Resh & Rosenberg, 1984) and have five tarsi (Hicken, 1967).

Adults live from a few weeks to several months, depending on species and nature of the habitat (Williams & Feltmate, 1992).

Males possess scent glands that lure the females into a swarm, generally over water. After gathering, a pair leaves the swarm from where they fly to the bank to copulate on river vegetation (Mackay & Wiggins, 1979). Females mate as soon as possible as they only live about five days (Hicken, 1967).

The eggs are laid in or near water, surrounded by a glutinous mass (Williams & Feltmate, 1992), which absorbs water and expands greatly after deposition. Female Trichoptera enter the water either by walking or by diving and glue the eggs to stones, or sometimes water plants, below the water surface (Mackay & Wiggins, 1979). The eggs hatch after about three weeks (Hicken, 1967).

Trichoptera inhabit a wide range of habitats from cool streams, showing the greatest diversity, to warmer streams, permanent lakes and marshes, and even permanent and temporary ponds. One species has been found in tidal pools off the coast of New Zealand (Williams & Feltmate, 1992).

Residents consider mass emergences of some species from large rivers a nuisance, as the scales on their wings can cause human allergies. The larvae of some Leptoceridae are reported to damage the young shoots of rice plants in paddy fields, while the larvae of a few species are known to eat fish eggs. On the beneficial side, many Hydropsychidae prey on Simuliidae larvae considered to be a pest and nuisance (Williams & Feltmate, 1992).

Furthermore, they are beneficially important in the trophic dynamics and energy flow in aquatic ecosystems and are food for many fish and water birds (Hicken, 1967). The larvae are also useful as biological indicator organisms for assessing water quality. Extensive studies of Trichoptera have been done for this purpose, because larvae of different species vary in sensitivity to various types of pollution (Resh & Unzicker, 1975, Resh, 1993, Dohet, 2002).

2.5.5 Megaloptera

The order Megaloptera is a small holometabolous order of insects in the infraclass Neoptera, division Endopterygota (Evans & Neunzig, 1996). The order consists of about 300 extant species (New & Theischinger, 1993). Some are very difficult to find in tropical climates (Henry *et al.* 1992), although they occur throughout most of the world. Megaloptera derived its name from the Latin meaning of "large wing" (Williams & Feltmate, 1992).

Megaloptera are divided into two families, the Corydalidae (fishflies & dobsonflies) and the Sialidae (alderflies) (Williams & Feltmate, 1992). Sialidae are classified in the guild of burrowers, while Corydalidae are generally seen as clingers or climbers (Williams & Feltmate, 1992).

Corydalidae associated with lentic habitats are found in well-oxygenated streams and lakes, as well as in productive ponds or swamps where dissolved oxygen may be very low. Sialidae associated with lotic environments occur in the same broad habitat categories, but are mostly confined to temperate latitudes, as well as usually requiring muddy or silty deposits and accumulated detritus (Williams & Feltmate 1992). The adults are poor fliers and therefore found around the ponds, lakes and streams which form the larval habitats (Evans & Neunzig, 1996).

The larvae of all species of Megaloptera are aquatic and attain the largest size of all aquatic insects (Kellogg, 1994). The larvae are elongate, moderately flattened, prognathous, with a distinct labrum and well developed mandibles. The abdominal appendages are used for identification and include lateral filaments on abdominal segments 1-8 in Corydalidae, while Sialidae bear them only on segment 1-7. Corydalidae also possess a pair of anal prolegs, compared to a single caudal filament in Sialidae. Members of the subfamily Corydalinae also possess tufts of accessory tracheal gills below the lateral filaments of segments 1-7 (Evans & Neunzig, 1996; Theischinger, 1991).

The larvae of Megaloptera are active predators, feeding on aquatic insects, annelids, crustaceans, and mollusks (Williams & Feltmate, 1992; Evans & Neunzig, 1996). Stewart *et al.* (1973) found larvae of Simuliidae (Diptera) and Hydropsychidae (Trichoptera:) as the main items in the diet of the Megaloptera *Corydalus cornutus*, while Hayashi (1988) found larvae of the Japanese Megaloptera, *Protohermes grandis* to feed on a wide variety of benthic macroinvertebrates including Ephemeroptera, Plecoptera, and Chironomidae (Diptera), showing some degree of cannibalism as well.

Megalopteran larvae go through 10-12 instars under natural conditions (New & Theischinger, 1993; Evans & Neunzig, 1996). This lifespan might take from one to three years in Sialidae (Azam & Anderson, 1969; Pritchard & Leischner, 1973; Hayashi, 1994), while Corydalidae have a lifespan between one and five years (Bowles, 1990; Williams & Feltmate, 1992).

When mature, the larvae constructs a pupal cell under large rocks on the stream bed (Williams & Feltmate, 1992) or leave the water and build a chamber under a rock or log, not too far from the aquatic habitat (New & Theischinger, 1993). Some have been reported to pupate as far as 50 m from the shore (Williams & Feltmate, 1992). In these chambers they spend several days as prepupae before they moult and become decticous and exarate pupae, and are capable of limited movement, including a strong defensive bite. After several more days (8-24 days in Corydalidae, 5-30 days in Sialidae), the adults emerge and the life cycle is completed (New & Theischinger, 1993).

Adult Megaloptera can be identified by the enlarged and fan-folded anal area of their hind wings (Borror *et al.*, 1989). Those of Corydalidae are large, pale yellowish to brownish or spotted black, head with ocelli, and their 4th tarsal segment is simple; while the wings of Sialidae are small, dark brown to grey and black, sometimes with orange spots on the head, head lacking ocelli, and their 4th tarsal segment is bilobed (Williams & Feltmate, 1992).

Adults are short-lived and generally do not feed, but they may drink water or sweet solutions (Evans & Neunzig, 1996). Mating and courtship behaviour are better known in Sialidae than in Corydalidae. Several studies on *Sialis* (New & Theischinger, 1993) reported reciprocal signaling, which implies a vertical vibration of the abdomen by males and females. Vibrations allow mutual recognition of species and sex. In some Corydalidae males may use female attractant scents from eversible glands near the genitalia (Contreras-Ramos 1998, 1999) as sexual communication. Corydalidae males fight when encountering each other (Parfin, 1952). Premating behavior in Corydalidae involves touching of antennae between male and female while facing one another, as well as male wing fluttering (Parfin, 1952).

After fertilization, the females lay elongate eggs in masses on vegetation overhanging the aquatic habitat or on large rocks projecting from the water. After about a week at cool temperatures, eggs hatch at night and first-instar larvae fall into the water (Williams & Feltmate, 1992). Sialidae egg masses are single layered, whereas Corydalidae lay their eggs in egg masses of one to five layers (Evans & Neunzig, 1996).

Megaloptera show some ecological value in that they prey on pestilent species of invertebrates, but none of the species are recognized as good bioindicator organisms (Mackie, 2001).

2.5.6 Diptera

The Diptera, or true flies, form a large order of endopterygote Neoptera. It is estimated that the order contains worldwide about 200 000 species, although only just over half of these have been described (Williams & Feltmate, 1992).

Approximately 32 families of Diptera contain species whose larvae are either aquatic or semi-aquatic. Eggs and pupae of these species are also aquatic, whereas adults are always terrestrial (Williams & Feltmate, 1992). Aquatic Diptera are divided in two suborders, the more primitive Nematocera (literally translated as "thread-like horn", and referring to the nature of the adult antenna), and the Brachycera ("short horn" meaning short antenna) (Williams & Feltmate, 1992). They can also be classified in the guilds of clingers, sprawlers, planktonic swimmers, burrowers, climbers, detritivores, or miners (Williams & Feltmate, 1992).

Diptera larvae occur in almost every conceivable aquatic habitat, from the bracts of pitcher plants, tree holes, saturated soil, mud puddles, to streams, ponds, large lakes, rivers, and even the marine rocky intertidal zone. Stratiomyidae have been recorded from geyser-fed thermal pools that reach temperatures up to 49^{o}C, and Ephydridae from natural seeps of crude petroleum. The only aquatic habitat where dipterans have not been recorded is the open ocean (Williams & Feltmate, 1992).

Diptera larvae are very diverse in shape and form, but all are united by the lack of any segmented thoracic legs (Foote, 1991) and elongated body (Merritt & Cummins, 1996). The head may be distinct, darkened and protruding anteriorly, reduced and partially retracted into the anterior thorax or reduced to no more than a remnant skeletal structure fully retained within the anterior thorax. The body may be smooth, or with a wide range of welts and tubercles. Legs, when present, are unsegmented prolegs, which may have a crown of crochets or claws (Foote, 1991).

The dietary diversity of aquatic Diptera larvae parallels their habitat diversity (Williams & Feltmate, 1992). A number of groups such as Simuliidae live in running water where they attach themselves to rocks or branches and use specially adapted labral fans to filter small particles from the water. Most Culicidae also feed by filtering the water column, but they are more typical in shallow, standing waters (Courtney *et al*., 1996). Other known Diptera families that fall in the guild of filter feeders are Chaoboridae, Dixidae, Blephariceridae and Ephydridae (Foote, 1991). Members of the families Syrphidae, Tipulidae, Psychodidae, Tabanidae and many species of Chironomidae feed on decaying organic matter and are known as shredders (Foote, 1991; Williams & Feltmate, 1992). Blephariceridae are specialized scrapers, consuming periphyton and associated materials from the surface of the substrate of fast-flowing streams. A number of families, including Chironomidae, Culicidae, Ceratopogonidae, Tabanidae, Empididae, Dolichopodidae and Muscidae have predatory species that feed on other insects and invertebrates either by engulfing the prey whole or piercing the skin and sucking out the body fluids (Foote, 1991; Courtney *et al*., 1996).

The number of larval instars varies in Diptera, with Nematocera having four and Brachycera mainly having three, but can have up to nine instars (Foote, 1991). Larval stages as short as several weeks and as long as two years have been reported, depending on the species, water temperature, and food conditions (Williams & Feltmate, 1992; Courtney *et al.,* 1996).

Amongst the aquatic Diptera, pupation may take place in the water or outside the aquatic environment in damp marginal habitats (Foote, 1991). The pupal stage lasts approximately two weeks, except in species that overwinter as pupae (Williams & Feltmate, 1992).

One pair of wings and a pair of small balancing halteres can be used for the identification of adult Diptera. Beyond this unifying feature there is a great diversity of body types. Those that do not feed as adults have reduced mouthparts, while groups that do feed possess highly specialized mouth parts (Merritt & Cummins, 1996), feeding on a wide variety of foods such as nectar, fungi, decaying vegetation, blood and flesh. Females of many species are anautogenous, requiring a blood meal to acquire sufficient protein to produce eggs (Ceratopogonidae, Culicidae, Simuliidae, Tabanidae)(Williams & Feltmate, 1992).

Mating of many Nematocera and some Brachycera occurs in swarms. After mating occurs, females leave the swarm to oviposit (Williams & Feltmate, 1992).

Egg-laying behaviour is diverse in the aquatic members with some scattering of eggs just below the surface on vegetation or on mineral substrates; others depositing eggs in gelatinous masses at, below, or above the water surface on emergent objects (Williams & Feltmate, 1992). The egg stage can last from a few days to a number of weeks (Courtney *et al*., 1996).

Most aquatic Diptera are univoltine, but under favourable conditions some may complete more than one generation a year. Some species may take two or three years to complete development in colder climates (Williams & Feltmate, 1992). The time required to complete the life cycle varies greatly among the species of this order and depends to a large extent on environmental conditions (Merritt & Cummins, 1996).

Aquatic Diptera represent some of the best-known insect forms, including Culicidae, Simuliidae, Chironomidae, Tipulidae and Syrphidae, many of which are the most troublesome of all insect pests, particularly in terms of human health and economics. Despite this, many groups of aquatic Diptera play crucial roles in the processing of food energy in aquatic environments and in supporting populations of fishes (Williams & Feltmate, 1992).

Chironomidae often the most abundant Diptera in aquatic systems (Ramsussen, 1985), are routinely used as test organisms in sediment toxicity test and bioassays (Chappie & Burton, 1997; Ibrahim *et al*. 1998; Conrad *et al.,* 1999). This is because they are relatively easy to breed, have a rapid lifecycle, and their sensitivity to many pollutants has been well documented (Ibrahim *et al*., 1998; Conrad *et al*., 1999; Ingersoll & Nelson, 1990; Crane *et al*., 2000).

2.5.7 Coleoptera

Of the more than one million described species of insect, at least one-third are beetles, making the Coleoptera (beetles) the most diverse and largest order of living organisms. It belongs to the infraclass Neoptera, division Endopterygota (Williams & Feltmate, 1992). Only about 5000 species have aquatic stages. Even so, the holometabolous Coleoptera are one of the most diverse freshwater macroinvertebrate groups. Eighteen families have species with at least one aquatic life stage (CSIRO, 1991; Williams & Feltmate, 1992; White & Brigham, 1996).

Aquatic Coleoptera species occur in the two major suborders, the Adephaga and the Polyphaga. Both larvae and adults of six beetle families are aquatic, Dytiscidae (predaceous diving beetles), Elmidae (riffle beetles), Gyrinidae (whirligig beetles), Haliplidae (crawling water beetles), Hydrophilidae (water scavenger beetles), and Noteridae (burrowing water beetles). Five families, Chrysomelidae (leaf beetles), Limnichidae (marsh-loving beetles), Psephenidae (water pennies), Ptilodactylidae (toe-winged beetles), and Scirtidae (marsh beetles) have aquatic larvae and terrestrial adults. Furthermore three families, Curculionidae (weevils), Dryopidae (long-toed water beetles), and Hydraenidae (moss beetles) have species that are terrestrial as larvae and aquatic as adults, a highly unusual combination among insects (Williams & Feltmate, 1992).

Aquatic Coleoptera can further be divided in the guilds clingers, climbers, sprawlers, swimmers, divers, and burrowers (Williams & Feltmate, 1992) and are found in all types of aquatic habitats but the diversity is highest in lentic water (White & Brigham, 1996).

Aquatic larvae are very variable, all with distinct sclerotized heads, strongly developed mandibles, 2-3 segmented antenna and three pairs of jointed thoracic legs, lacking any abdominal prolegs (CSIRO, 1991).

Coleoptera larvae can be herbivores (chewers or piercers), scavengers (gathering collectors), or voracious predators (engulfers or piercers). Dytiscidae larvae are well known for their piercing mandibles, with which they inject proteolytic enzymes into their prey, resulting in subsequent ingestion of internal tissues (Williams & Feltmate, 1992). Other predatory Coleoptera either engulf their prey or grasp the prey with fang-like mandibles (White & Brigham, 1996).

The Coleoptera larvae undergo between three to eight moults (Williams & Feltmate, 1992) taking up to a month to complete development (White & Brigham, 1996).

The pupal phase of all Coleoptera is technically terrestrial and prior to pupation the final instar larvae have to crawl out of the water and dig a pupal chamber in litter along the shore (White & Brigham, 1996). A few species have diapausing prepupae, but most complete transformation to adults in two to three weeks (Williams & Feltmate, 1992).

The beetles as adults are easily identified by an anterior pair of harden and leathery elytra and not used in flight, whereas the membranous hind wings, which are used for flight, are concealed under the elytra when the animals are at rest (Williams &

Feltmate, 1992). The size of the body can range from 2-30 mm depending on family and species. The larvae are much more variable in form. Some have large fang-like mandibles, while others have chewing mouthparts. The abdomen can have lateral projections on it or dorsal ornamentation (White & Brigham, 1996).

Most species are detritivores or predators, but some species have been reported to scrape blue-green algae from substrates (Williams & Feltmate, 1992).
Terrestrial adults of aquatic beetles are typically short-lived and sometimes non-feeding, like those of the other orders of aquatic insects (Williams & Feltmate, 1992).

Eggs are laid in spring by over wintering females (White & Brigham, 1996) and hatch in one or two weeks, with diapause occurring rarely (Williams & Feltmate, 1992)

Most larval and adult beetles are tolerant of wide changes in pH and dissolved oxygen, making Coleoptera less suitable as bioindicators. Therefore only a few Coleoptera are recognized as indicator organisms of environmental health. Their main indicator value is in the physical type of habitat they utilize (Mackie, 2001).

2.5.8 Hemiptera

The Hemiptera (true bugs) is a widespread and specious order (Muller, 1982), belonging to the infraclass Neoptera, in the division Exopterygota; their wings develop externally and can be folded over the dorsum. Only about 10% of all species of Hemiptera are associated with aquatic environments, and these are representatives of 15 families of the suborder Heteroptera (Williams & Feltmate, 1992). The Homoptera suborder will not be considered here, as its only aquatic members are a few semi-aquatic aphids (McCafferty, 1981). The aquatic Heteroptera can be divided into six fully aquatic families that are aquatic in all life stages (Belostomatidae, Corixidae, Naucoridae, Nepidae, Notonectidae, & Pleidae). These families are classified as swimmers, clingers, or climbers (Williams & Feltmate, 1992); four families that are surface dwellers or skaters (Gerridae, Hydrometridae, Mesoveliidae & Veliidae) and are thus semi-aquatic (Williams & Feltmate, 1992); and four families that are shore bugs and live along the edges of ponds or streams (Gelastocoridae, Hebridae, Ochteridae & Saldidae) and can be classified in guilds as skaters, climbers, clingers, burrowers, or sprawlers (Williams & Feltmate, 1992).

The most obvious feature of all Hemiptera is their suctorial mouthparts, which consist of hinged stylets (mandibles & maxillae) resting in an anteriorly grooved rostate labium (Daly *et al.*, 1997). Identification to family level can be based on body proportions, shape of head and prothorax, leg configuration and other readily-visible external features (Muller, 1982).

Hemipterans are paurometabolous, undergoing incomplete, gradual metamorphosis from egg to larvae to adult. Eggs hatch after one to four weeks of embryonic development. The larvae undergo five moults before moulting to the sexually mature adults. Hemipterans generally over winter as adults, and in regions where ponds freeze, many species fly to streams to over winter. In temperate regions, life cycles are usually univoltine, or occasionally bivoltine, with reproduction occurring in the summer. Multivoltinism is more common in the tropics (Williams & Feltmate, 1992).

In common with other hemimetabolous insects, the nymphs are very similar to the adults in terms of their appearance, habitat and behaviour (Williams & Feltmate, 1992), except for being smaller in the a larval stage and in the presence of wings in the adult. However, some adult forms of the families Gerridae and Vellidae may be apterans (Muller, 1982; Polhemus, 1996).

Most members of the common family Corixidae feed on plant material. Other families are primarily predatory on aquatic invertebrates. Some of the larger species such as *Lethocerus americanus*, will feed on small vertebrates. Their forelegs are used to grasp the prey and the beak injects digestive juices to kill the prey. The internal tissues of the prey are liquefied by the digestive juices and sucked out (Polhemus, 1996).

Aquatic Hemiptera hold an important place in the ecology of freshwater systems. They are important food for many organisms (Clark, 1992; McCafferty, 1981). These insects generally hold an intermediate place in food chains, and apart from being eaten, are often important predators (Runck & Blinn, 1994). Another important function of aquatic Hemipterans in ecosystems is to inhabit conditions that would be extremely stressful for other organisms. Many of these insects have the ability to live in extremely adverse environments. Corixidae, in particular, are some of the first colonizers of extreme environments. These insects have been found in German mining lakes with a pH below 3 (Wollmann, 2001). They have also been present in Finland waters heavily polluted by sewage (Jansson, 1987). In such environments, aquatic Hemipterans may be the first step in redeveloping a healthy ecosystem.

As could be expected from their important role in ecosystems, this group of insects also has much economic importance. They control Culicidae populations, provide food for fish (McCafferty, 1981), are used as bioindicators (Jansson, 1987; Papacek, 2001; Wollmann, 2001), and even provide food for people in some countries (Glausiusz, 2004). The economic importance of these insects has probably been underestimated (Papacek, 2001).

On the negative side aquatic Hemiptera do not make great bioindicators, due to their sustainability to variation in environmental conditions and life style that do not depend on water quality (Mackie, 2001). The predatory species are also occasional pests in manmade fish hatcheries where they feed off of the young fish (McCafferty, 1981).

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Chapter Three

MATERIALS AND METHODS

3.1 SITES

A total of 30 sites were chosen on feasibility of access in winter, when water levels were the highest, along the Eerste River at Jonkershoek, Western Cape, South Africa (Fig. 3.1.1). Fifteen sites downstream from the Klein Plaas dam at the Yellow foot trail station (referred to here as “downstream”) and fifteen sites upstream from the Klein Plaas dam (referred to here as “upstream”), with nine at the Concrete bridge station and six at the White bridge station, were selected. At every site, the location (Table 3.1) and elevation was measured using a Global Positioning System and marked with a clearly visible white plastic band. The three sampling stations will be referred to as, the “Yellow foot station”, “Concrete bridge station” and the “White bridge station” respectively. Further divisions relate to specific sites.

3.2 SAMPLING

3.2.1 Sampling procedure

Samples were obtained using the SASS5 kick-sampling method, where rocks and other benthic material were disturbed to flow downstream into a soft, 1mm mesh net, 30 cm in diameter. This was done in all possible microhabitats within any one site for 15 min.

The content of each sample was then washed down to the bottom of the net and carefully tipped into a tray by inverting the net. The net was then flushed out with water to transfer all biota to the tray. Samples were then placed in 96% ethanol and taken to the laboratory for further analysis.

Figure 3.1.1. Locality map of the Western Cape nature conservation areas, indicating the study area at no.9 (enlarged in the smaller map).

Table 3.1.1 Co-ordinates and sample numbers of sites used along the Eerste River catchment.

	Site	Area	Latitude	Longitude	6-8 October 2003	16-18 February 2004	1-3 March 2004	15-17 March 2004	29-31 March 2004	10-12 May 2004	21-23 June 2004
Downstream	1	Yellow foot station	33°58'11.6"S	18°55'55.9"E	B1	D1	E1	F1	G1	H1	I1
	2	Yellow foot station	33°58'12.5"S	18°55'55.5"E	B2	D2	E2	F2	G2	H2	I2
	3	Yellow foot station	33°58'12.9"S	18°55'55.6"E	B3	D3	E3	F3	G3	H3	I3
	4	Yellow foot station	33°58'13.0"S	18°55'56.1"E	B4	D4	E4	F4	G4	H4	I4
	5	Yellow foot station	33°58'14.6"S	18°55'55.6"E	B5	D5	E5	F5	G5	H5	I5
	6	Yellow foot station	33°58'14.6"S	18°55'55.5"E	B6	D6	E6	F6	G6	H6	I6
	7	Yellow foot station	33°58'16.6"S	18°55'57.1"E	B7	D7	E7	F7	G7	H7	I7
	8	Yellow foot station	33°58'16.7"S	18°55'57.1"E	B8	D8	E8	F8	G8	H8	I8
	9	Yellow foot station	33°58'18.0"S	18°55'59.2"E	B9	D9	E9	F9	G9	H9	I9
	10	Yellow foot station	33°58'18.6"S	18°55'59.2"E	B10	D10	E10	F10	G10	H10	I10
	11	Yellow foot station	33°58'19.6"S	18°55'59.0"E	B11	D11	E11	F11	G11	H11	I11
	12	Yellow foot station	33°58'20.2"S	18°56'01.2"E	B12	D12	E12	F12	G12	H12	I12
	13	Yellow foot station	33°58'21.0"S	18°56'02.0"E	B13	D13	E13	F13	G13	H13	I13
	14	Yellow foot station	33°58'20.9"S	18°56'01.9"E	B14	D14	E14	F14	G14	H14	I14
	15	Yellow foot station	33°58'21.6"S	18°56'04.2"E	B15	D15	E15	F15	G15	H15	I15
		Klein Plaas Dam	33°58'42.0"S	18°56'36.0"E							
Upstream	16	Concrete bridge station	33°58'56.5"S	18°57'06.2"E	B29	D29	E24	F24	G24	H24	I24
	17	Concrete bridge station	33°58'57.6"S	18°57'06.7"E	B28	D28	E23	F23	G23	H23	I23
	18	Concrete bridge station	33°58'58.6"S	18°57'06.9"E	B27	D27	E22	F22	G22	H22	I22
	19	Concrete bridge station	33°58'59.3"S	18°57'07.7"E	B26	D26	E21	F21	G21	H21	I21
	20	Concrete bridge station	33°59'00.4"S	18°57'08.6"E	B25	D25	E20	F20	G20	H20	I20
	21	Concrete bridge station	33°59'01.1"S	18°57'08.8"E	B24	D24	E19	F19	G19	H19	I19
	22	Concrete bridge station	33°59'01.9"S	18°57'08.7"E	B23	D23	E18	F18	G18	H18	I18
	23	Concrete bridge station	33°59'02.5"S	18°57'09.0"E	B22	D22	E17	F17	G17	H17	I17
	24	Concrete bridge station	33°59'05.0"S	18°57'11.0"E	B21	D21	E16	F16	G16	H16	I16
	25	White Bridge station	33°59'40.1"S	18°57'30.7"E	B31	D30	E25	F25	G25	H25	I25
	26	White Bridge station	33°59'38.0"S	18°55'31.7"E	B32	D31	E26	F26	G26	H26	I26
	27	White Bridge station	33°59'37.5"S	18°55'32.2"E	B33	D32	E27	F27	G27	H27	I27
	28	White Bridge station	33°59'37.2"S	18°55'33.1"E	B34	D33	E28	F28	G28	H28	I28
	29	White Bridge station	33°59'36.8"S	18°55'33.9"E	B35	D34	E29	F29	G29	H29	I29
	30	White Bridge station	33°59'36.6"S	18°55'33.9"E	B36	D35	E30	F30	G30	H30	I30

Samples were obtained on seven different occasions, 6-8 October 2003, 16-18 February 2004, 1-3 March 2004, 15-17 March 2004, 29-31 March 2004, 10-12 May 2004 and 21-23 June 2004.

3.2.2 Physical data

On each sampling occasion, the water's physical state was measured using a multi-probe system, for pH, temperature, conductivity, dissolved oxygen in percentage as well as $mg.\ell^{-1}$. Furthermore, the percentage cover and stream flow was also determined for each site.

3.3 LABORATORY WORK

All samples were then cleaned and all individuals identified to family level, using the dichotomous keys of Gerber & Gabriel (2002) and counted. The sorted individuals were stored in 75% ethanol.

3.4 DATA ANALYSES

All calculations, statistical analyses and graphs were calculated and drawn up by using Microsoft Excel for Windows, except where indicated otherwise.

3.4.1 Invertebrate presence

The percentages of macroinvertebrate taxa in the Eerste River for each of the sampling periods, as well as for the combined data were calculated. The percentages were used to indicate the differences in taxa present, upstream and downstream from the Klein Plaas dam.

3.4.2 South African Scoring System (SASS5)

SASS5 values according to Dickens & Graham (2002) were allocated for each taxon per sample. These values were then added up to calculate the SASS5 score. The Average Score per Taxon (ASPT) was determined by dividing the SASS5 scores by the number of taxa at each site. These values, for each of the sampling periods, were used to correlate SASS5 scores as well as ASPT scores of each site.

3.4.3 Physical data

A single factor ANOVA was done on the data from upstream and downstream areas for temperature, conductivity, dissolved oxygen, percentage dissolved oxygen, pH, SASS5 scores, ASPT scores and number of individuals for each sampling period, as well as for the combined data.

3.4.4 Bray-Curtis similarity

Sites were scored by hand for presence (1) or absence (0) of each family to create binary matrices from which a Bray-Curtis similarity chart was drawn up with the program, Primer 5 for Windows Version 5.2.9. Using these similarity values, dendrograms for each of the sampling periods, as well as one for the combined data, were created.

3.4.5 Simpson's index of diversity

Diversity was calculated for each of the samples using the Simpson's index of diversity (D_s)(Stilling, 1992),

$$D_s=\Sigma\ (n_i(n_i-1))/(N(N-1))$$

where n is the number of individuals of a specific taxa and N is the total number of individuals.

3.4.6 Shade evaluation

The mean temperature, conductivity, dissolved oxygen, percentage dissolved Oxygen, pH, SASS5 scores, ASPT scores, number of individuals and Simon's index of diversity for each of the different shading categories (less than 33%, between 33- 66% and more than 66%) were calculated and used to indicate the differences between each sampling period.

3.4.7 Regression analysis

Regression analysis was calculated between the number of taxa (dependent) and each of the variables, temperature, conductivity, dissolved oxygen, percentage dissolved oxygen as well as pH (independent).

Further regression analysis was done on the number of individuals (dependent) for each of the variables, temperature, conductivity, dissolved oxygen, percentage dissolved oxygen as well as pH (independent).

3.5 REFERENCES

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Chapter Four

RESULTS

4.1 INVERTEBRATE PRESENCE

During the study period (October 2003 – June 2004), a total of 20 760 invertebrate individuals were sampled and identified, mainly in five insect orders in the Eerste River system. Ephemeroptera (48%), Diptera (24%), Coleoptera (10%), Trichoptera (7%) and Plecoptera (5%), with all other taxa comprising only 6% (Fig4.1.1).

Figure 4.1.2.1 indicates the influence of season on the numbers of individuals in each order, with Ephemeroptera as the dominant taxon. Their abundance was highest during late winter to early spring, with numbers decreasing sharply in the summer months. From autumn to winter their numbers increased. Plecoptera and Coleoptera abundance remained constant with a small increase in abundance during autumn for Plecoptera and in summer for Coleoptera. Diptera also showed a small decrease in abundance over summer, but reached highest numbers in late autumn, then decreased during winter. Trichoptera showed a steady increase from winter to autumn, then a steady decline towards winter.

The rare taxa (Fig. 4.1.2.2), like the Odonata, Hemiptera, Megaloptera, Crustacea and Gastropoda, showed a summer peak and winter low. Turbularia and Hidracarina, numbers were extremely low in spring, but highest late autumn for Turbularia and winter for Hidracarina. Annelida abundance was constant, with a small increase early winter.

Whether samples were taken downstream or upstream of Klein Plaas dam influenced these patterns.

Ephemeroptera numbers (Fig. 4.1.3.1) were much higher upstream than downstream (a difference of 705.3 individuals between their means). It was clear that the upstream population that influenced the patterns as seen above (Fig. 4.1.1), as the downstream population's abundance did not vary significantly.

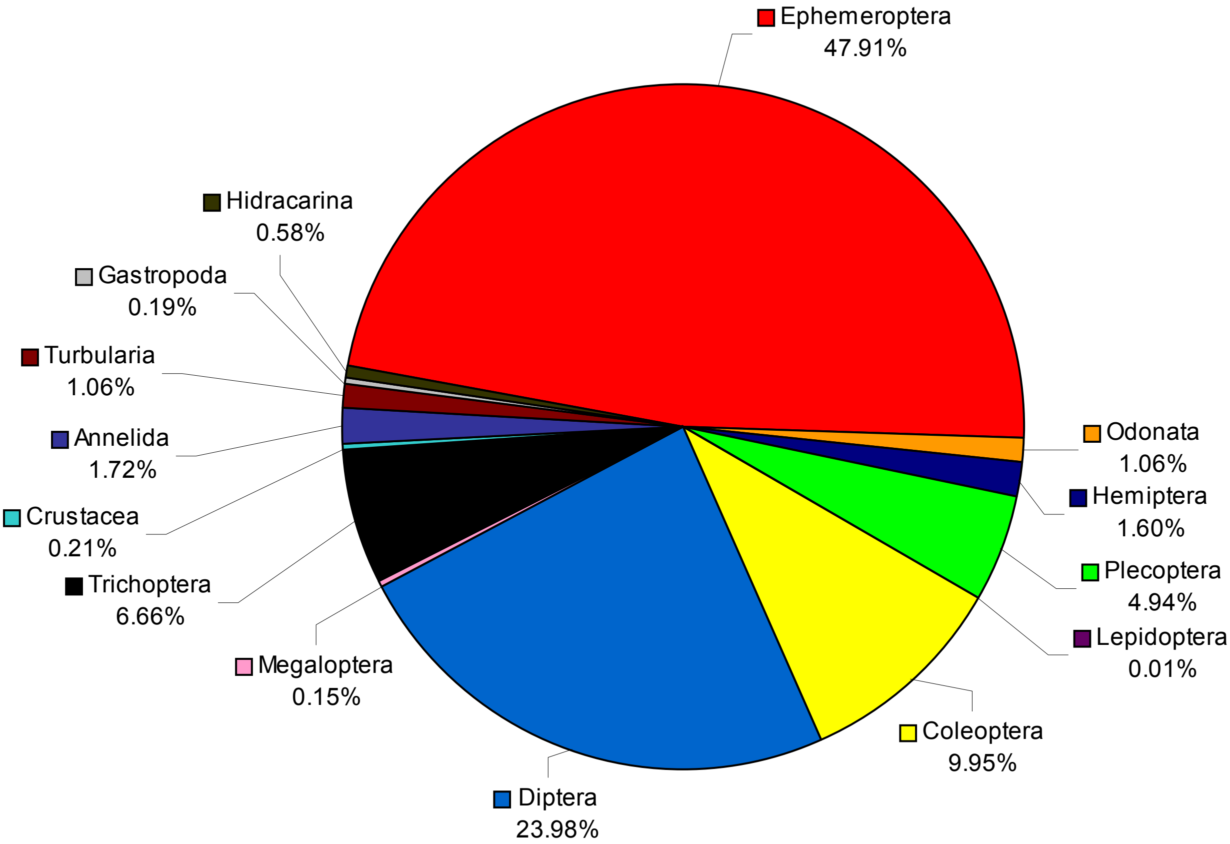

Figure 4.1.1 Percentage of macroinvertebrate taxa found during the sampling period, October 2003 - June 2004, in the Eerste River, Jonkershoek.

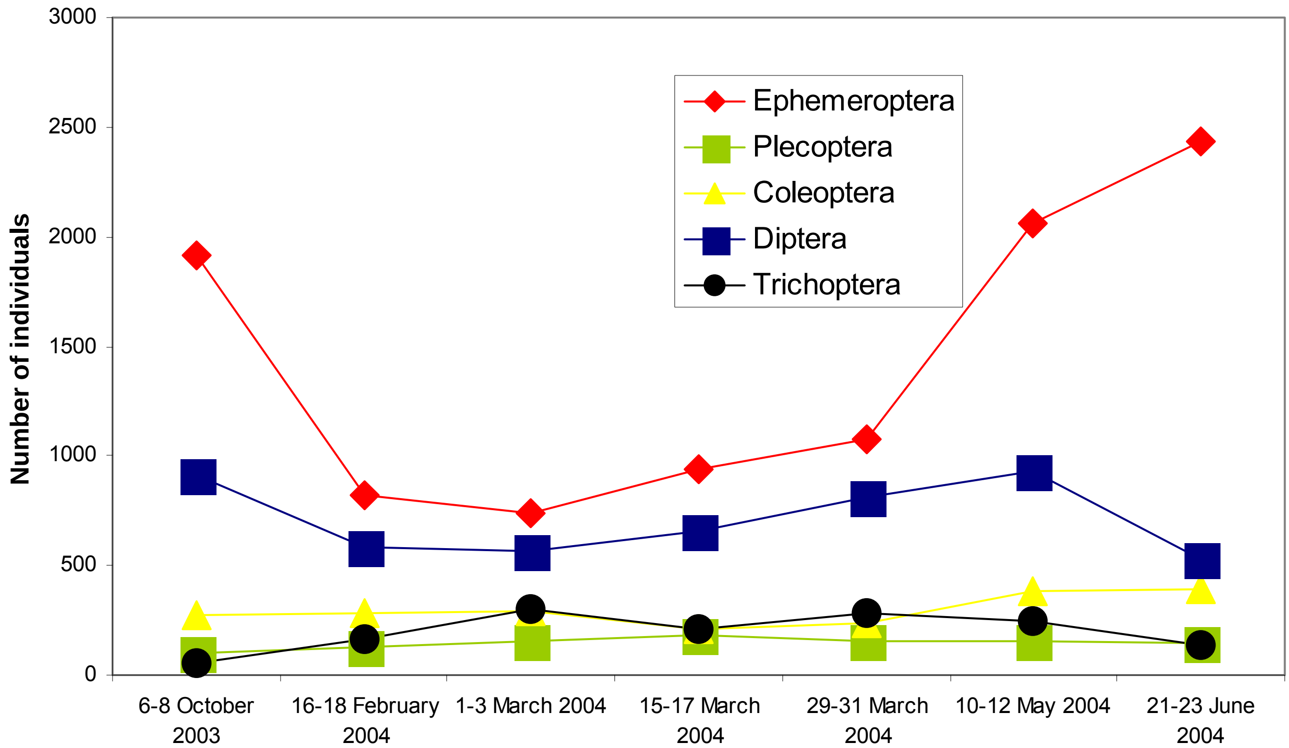

Figure 4.1.2.1 Presence of the dominant invertebrate taxa during the sampling period, October 2003 - June 2004, in the Eerste River, Jonkershoek.

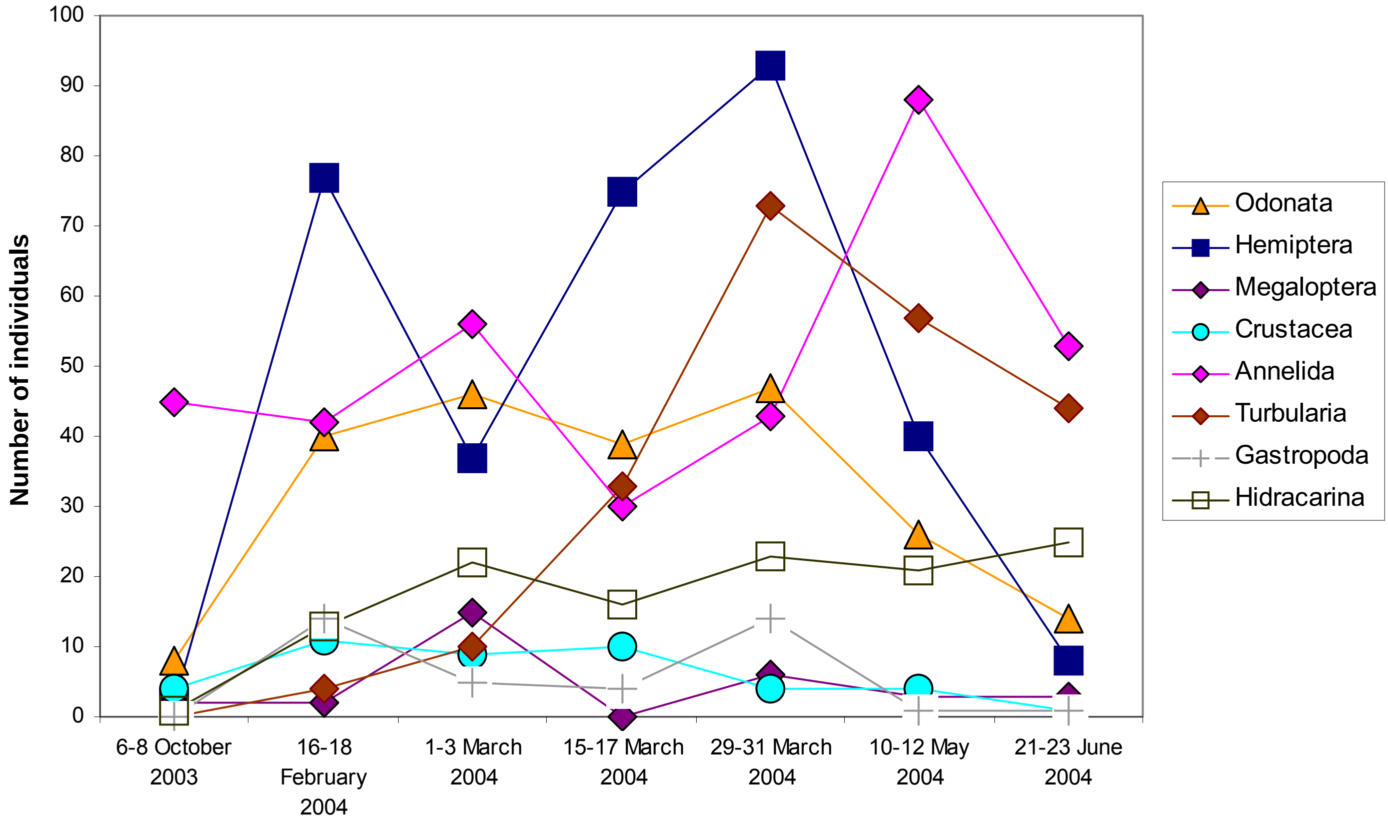

Figure 4.1.2.2. Presence of the less dominant invertebrate taxa during the sampling period, October 2003 - June 2004, in the Eerste River, Jonkershoek.

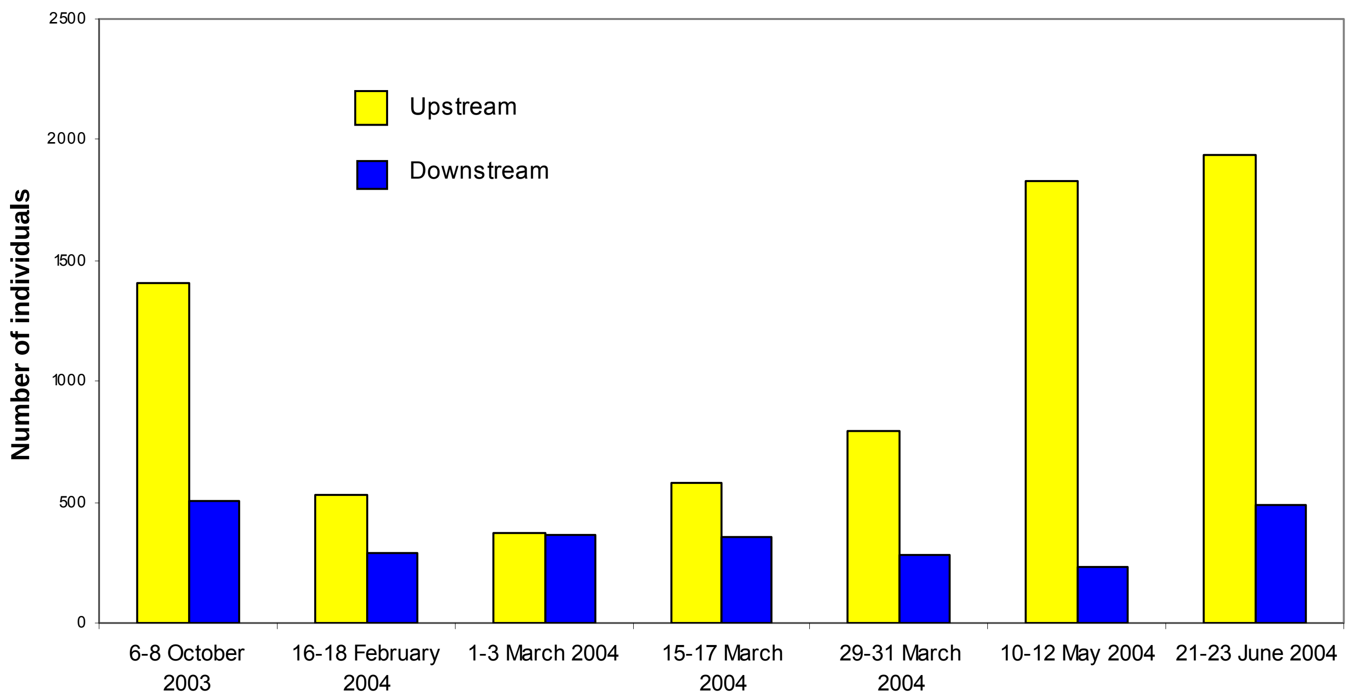

Figure 4.1.3.1. Differences in Ephemeroptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

The Diptera numbers (Fig. 4.1.3.2) in the upstream sampling area peaked in autumn, while the downstream sampling area showed an opposite trend in that abundance, decreasing over summer, peaking in spring and early winter, and decreasing again during winter. In all sampling periods, the upstream sampling areas had greater numbers except for spring and early winter, where the opposite was true. During the winter sampling periods, the abundance are so much greater that they influence the total, so that the downstream sampling area is the richer in Diptera than the upstream sampling area, with a mean difference of 32.2 individuals per sampling period.

The upstream sampling area has a much greater abundance of Coleoptera (Fig. 4.1.3.3), with the mean per sample period varying by 262 individuals. In both cases there was an increase in the abundance towards winter, with the upstream sampling area showing a slight drop during late autumn.

There was a great difference in Trichoptera abundance (Fig. 4.1.3.4) between the two sampling areas, with the downstream sampling area having a greater abundance in individuals than the upstream area. The only exception was in spring, when the upstream sampling area had a higher abundance than the downstream area. The total number of Trichoptera sampled during the study period differed by 720 individuals between the two sampling areas, indicating a mean difference of 102.8 individuals per sampling period. The downstream sampling area had a bell shape curve with its peak in autumn. The upstream sampling area had a steady increase in numbers towards winter, with the highest number of individuals in early winter (83 individuals).

There was no significant difference between the two sampling areas for Odonata (Fig. 4.1.3.5). The data indicated a small difference of less than one between the means per sample period with both areas having a bell-shaped curve that peaks in autumn.

There were no changes in the pattern of abundance in Hemiptera (Fig. 4.1.3.6) with time of the year, but there were differences in the number of individuals between upstream and downstream. The mean difference is 21.2 individuals in favour of the downstream sampling area.

Megaloptera abundance (Fig. 4.1.3.7) overall was very low, with only 31 individuals of Corydalidae during the whole sampling period. Of these individuals, only two were

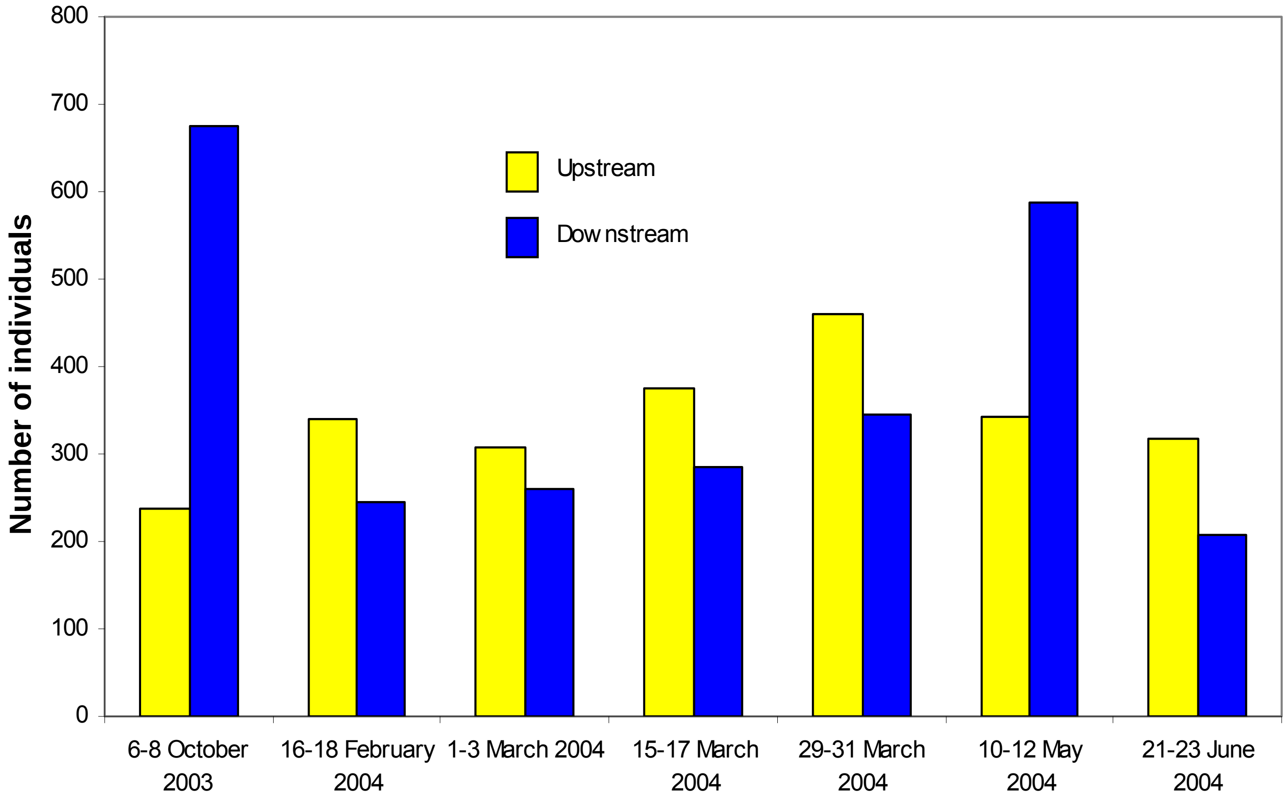

Figure 4.1.3.2. Differences in Diptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

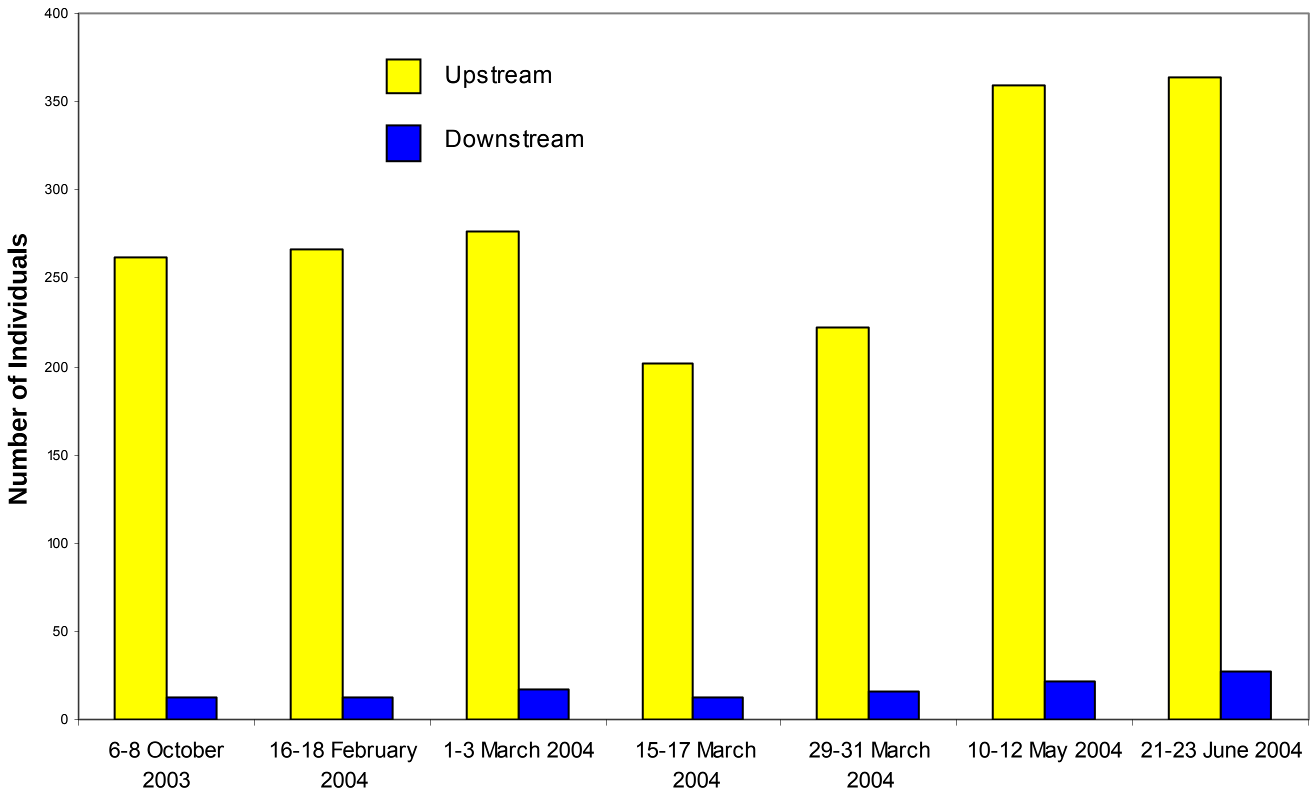

Figure 4.1.3.3. Differences in Coleoptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

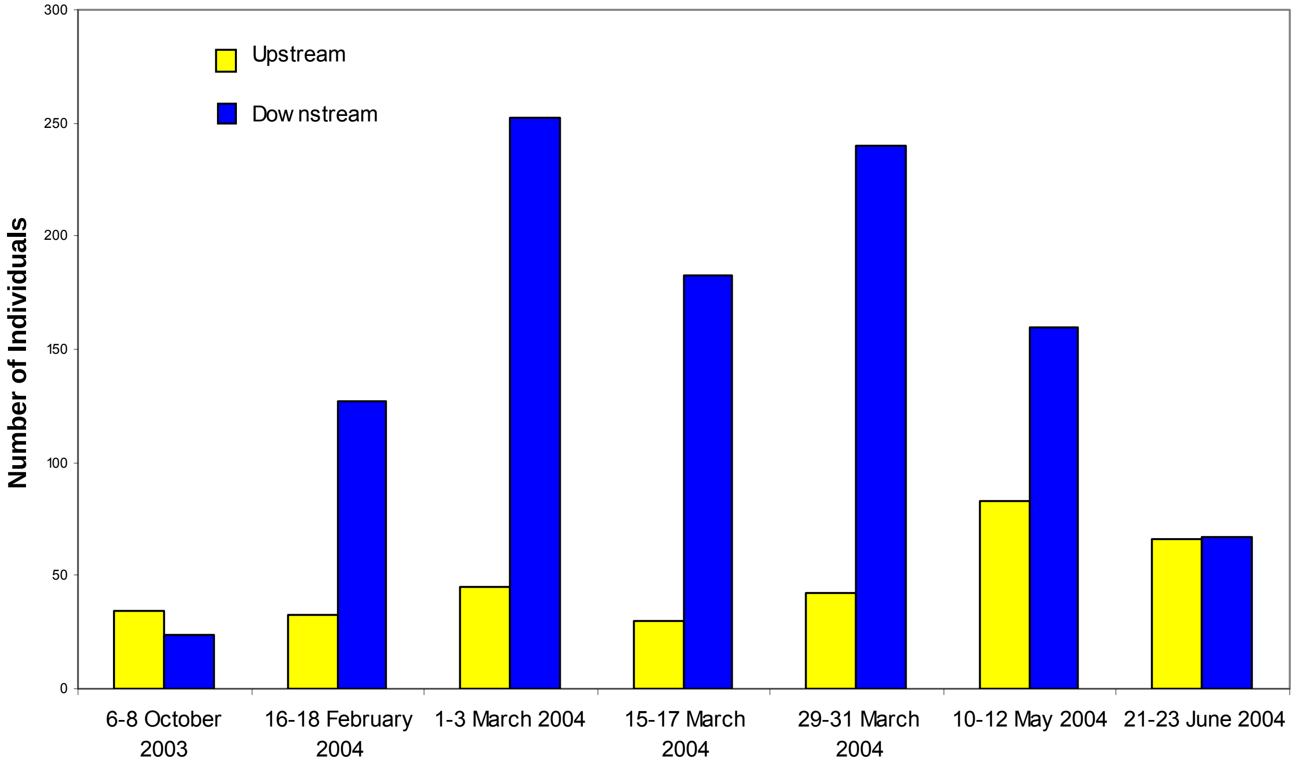

Figure 4.1.3.4. Differences in Trichoptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

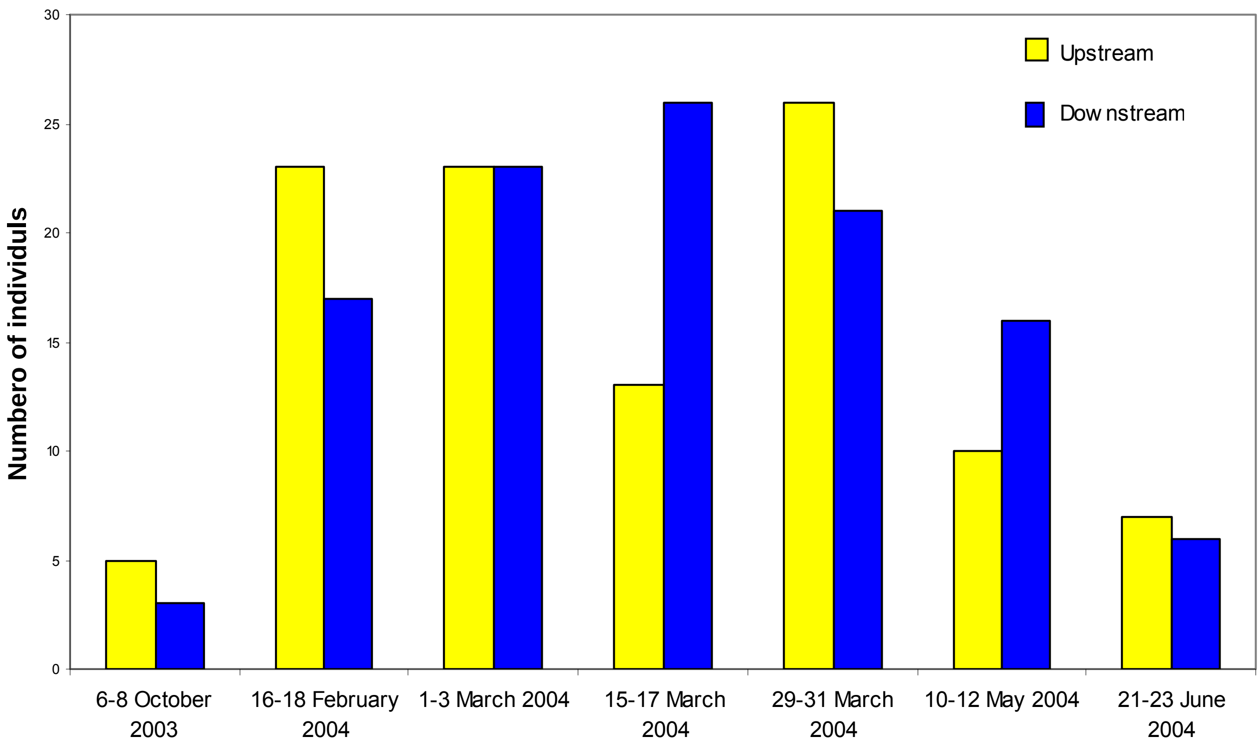

Figure 4.1.3.5. Differences in Odonata numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

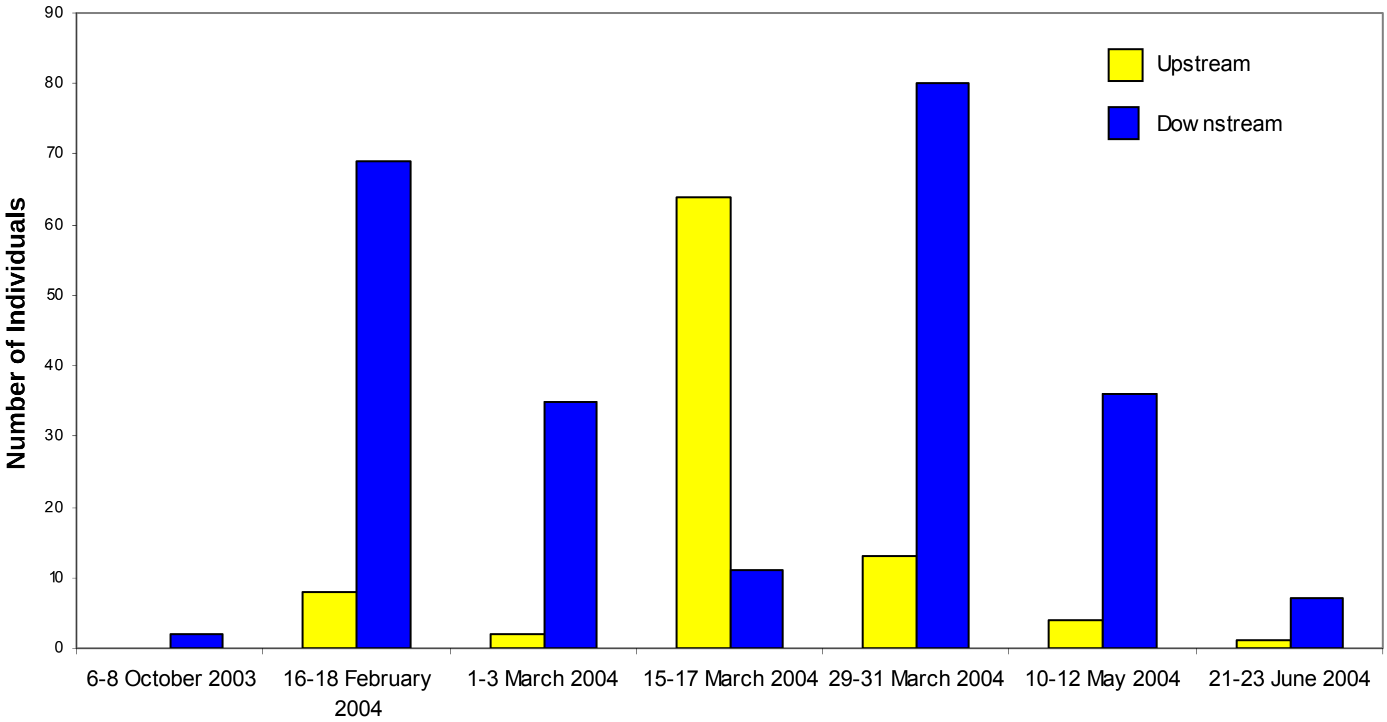

Figure 4.1.3.6. Differences in Hemiptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

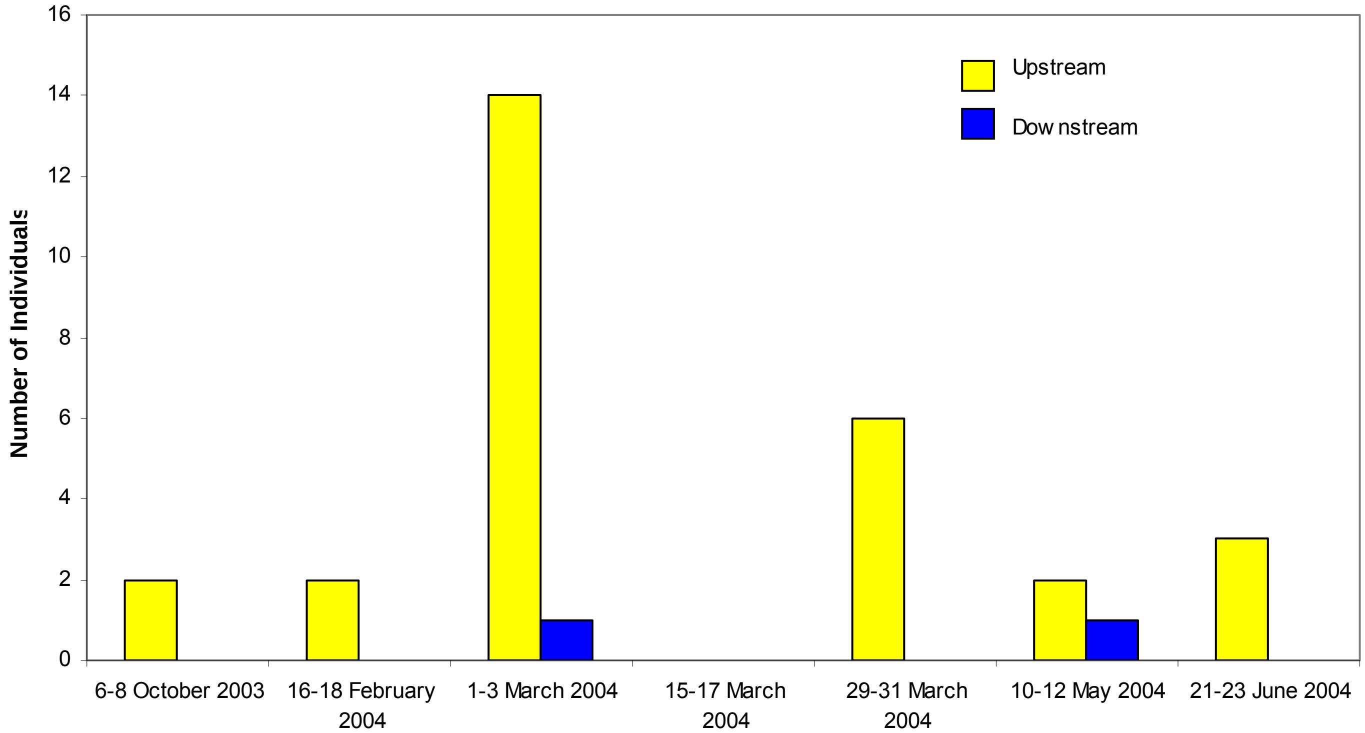

Figure 4.1.3.7. Differences in Megaloptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

downstream, giving a difference in mean number of individuals of 3.8 between the two sampling areas. There was also no distinct pattern in distribution over time (the maximum and minimum numbers both were in March).

As with the Ephemeroptera, there was a clear difference in Plecoptera numbers (Fig. 4.1.3.8), between the two sampling areas. Plecoptera numbers upstream had a total greater than 1000, yet downstream there were only 11 individuals, a difference in mean number of individuals per sample period of 143.8. Individuals upstream showed a slight curve that peaked in autumn, with the lowest values in spring.

Annelida abundance (Fig. 4.1.3.9) was high downstream, with a difference between upstream and downstream of 23.6 individuals per sample period. In both cases there was a similar trend, with a steady increase in numbers towards winter. In both instances, there was a small decline in abundance in mid- and late-March.

Turbularia abundance (Fig. 4.1.3.10) upstream and downstream had the same trend, with a steady increase in abundance towards early winter, decreasing in winter. None were recorded in spring. In all sampling periods except February, downstream had a higher individual count of Turbularia than upstream, giving a mean difference 5.8 individuals per sampling period.

Hidracarina abundance (Fig. 4.1.3.11) differed between sampling areas, with an upstream bell-shaped curve peaking in autumn, and a downstream one peaking in winter. In summer the population upstream was higher than the downstream population but for the rest of the sampling periods the opposite was true, with a difference of 7.9 individuals per sample.

The upstream Crustacea population peaked in summer and early autumn, while downstream it decreased from its highest point in spring to its lowest in winter (Fig. 4.1.3.12). There where no crustaceans upstream during spring and late winter. The abundance between the two sampling areas did not vary significantly, with a mean difference of only 1.3 individuals per sampling period.

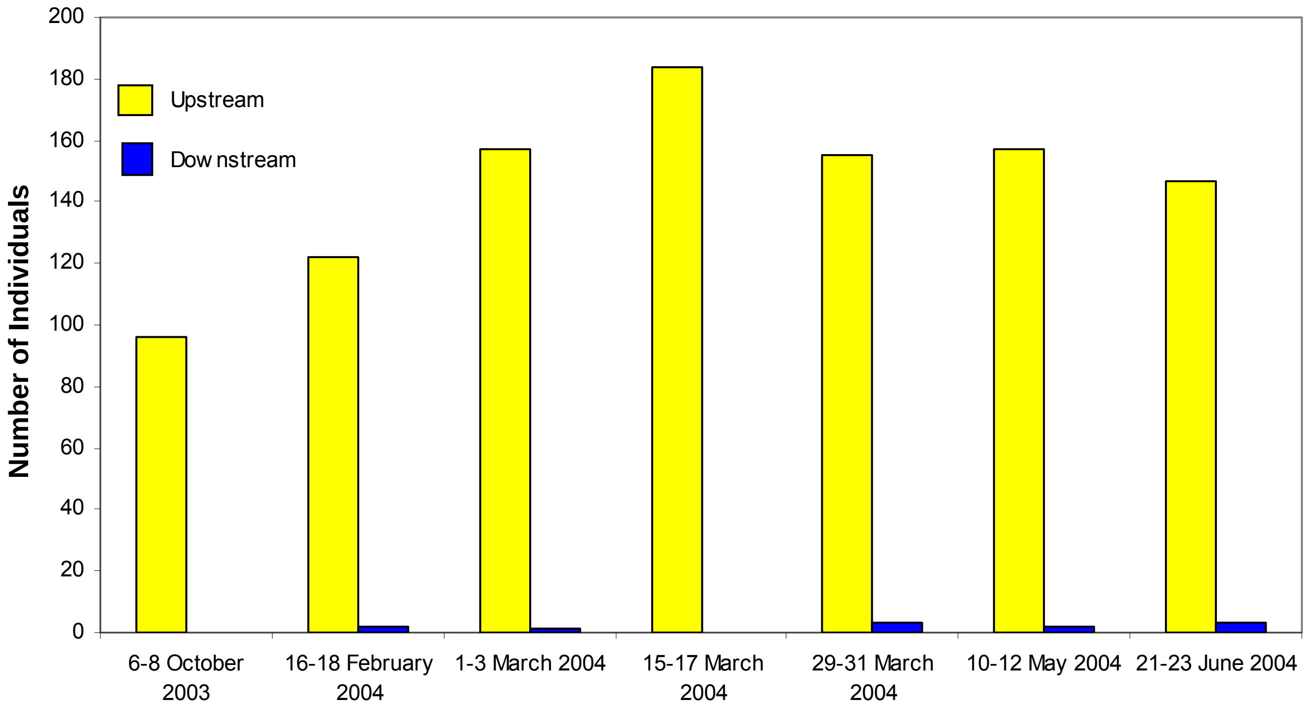

Figure 4.1.3.8. Differences in Plecoptera numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

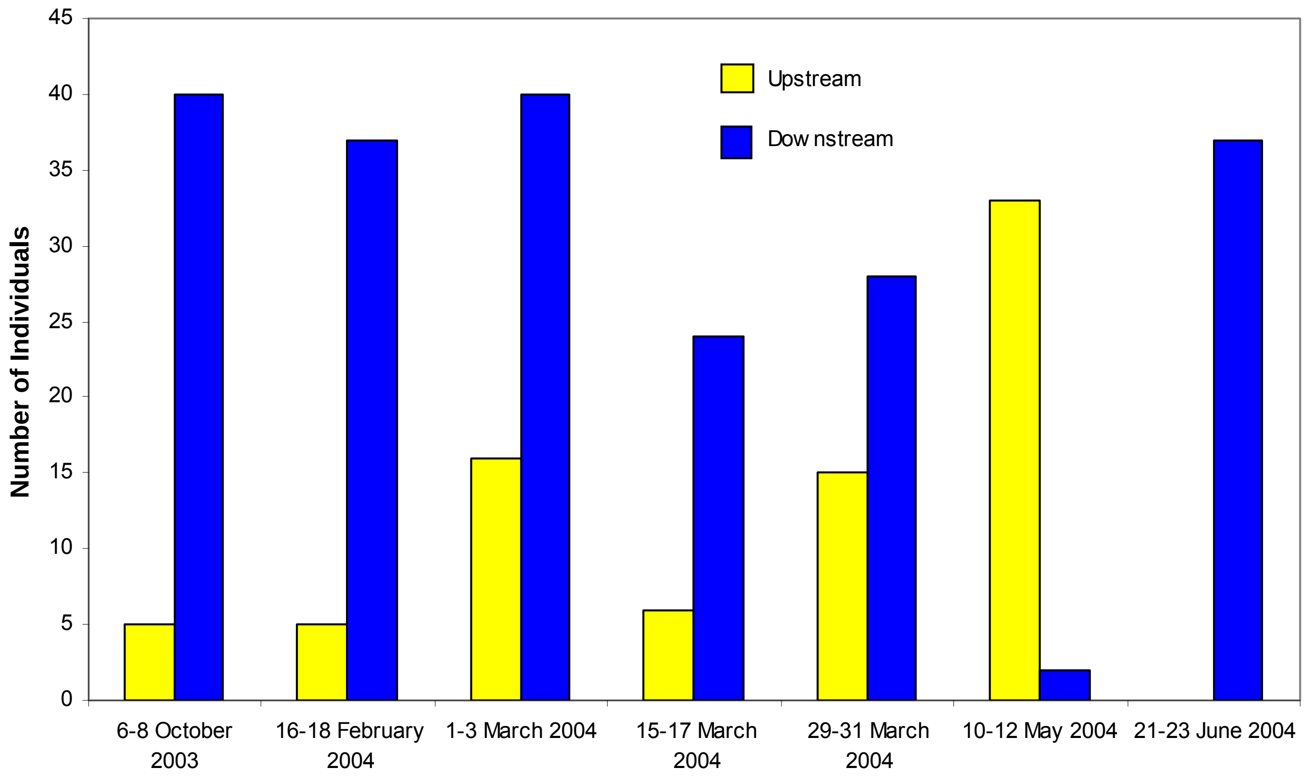

Figure 4.1.3.9. Differences in Annelida numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

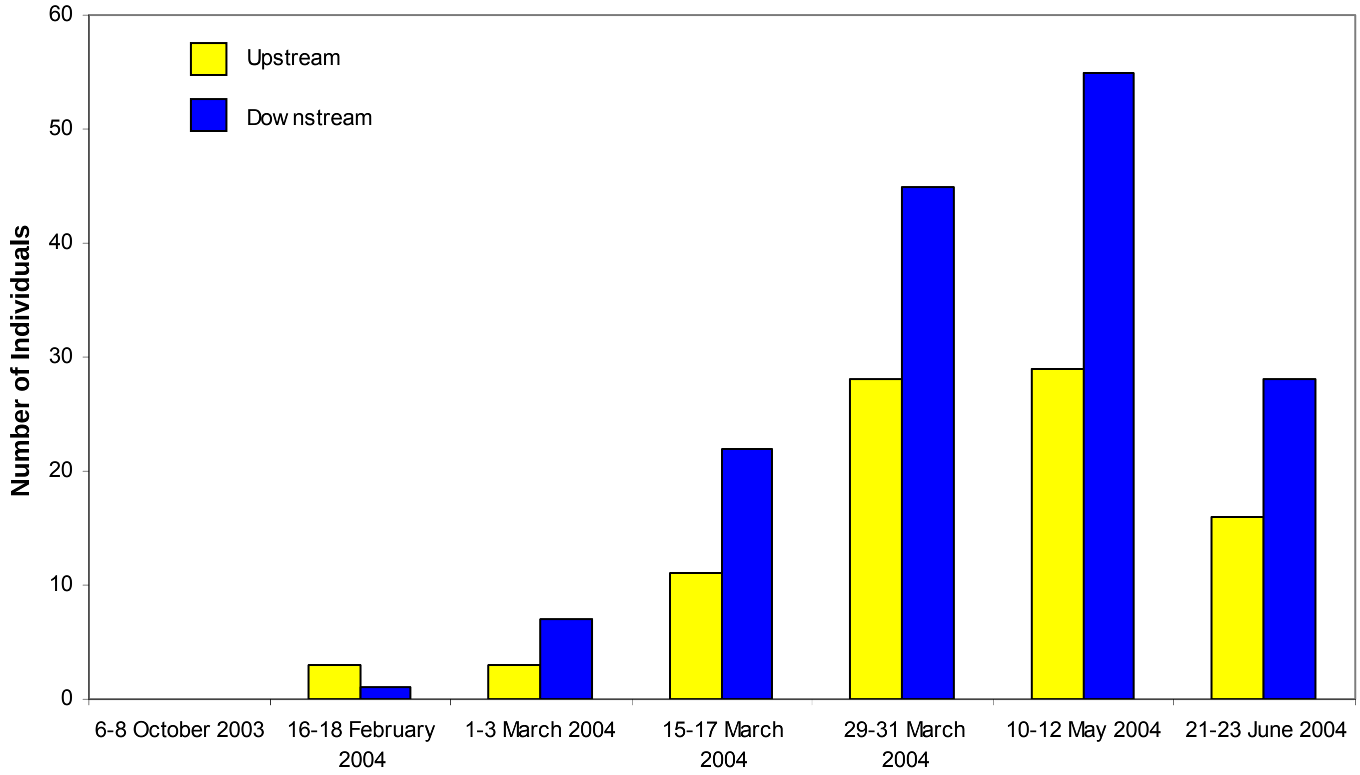

Figure 4.1.3.10. Differences in Turbularia numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

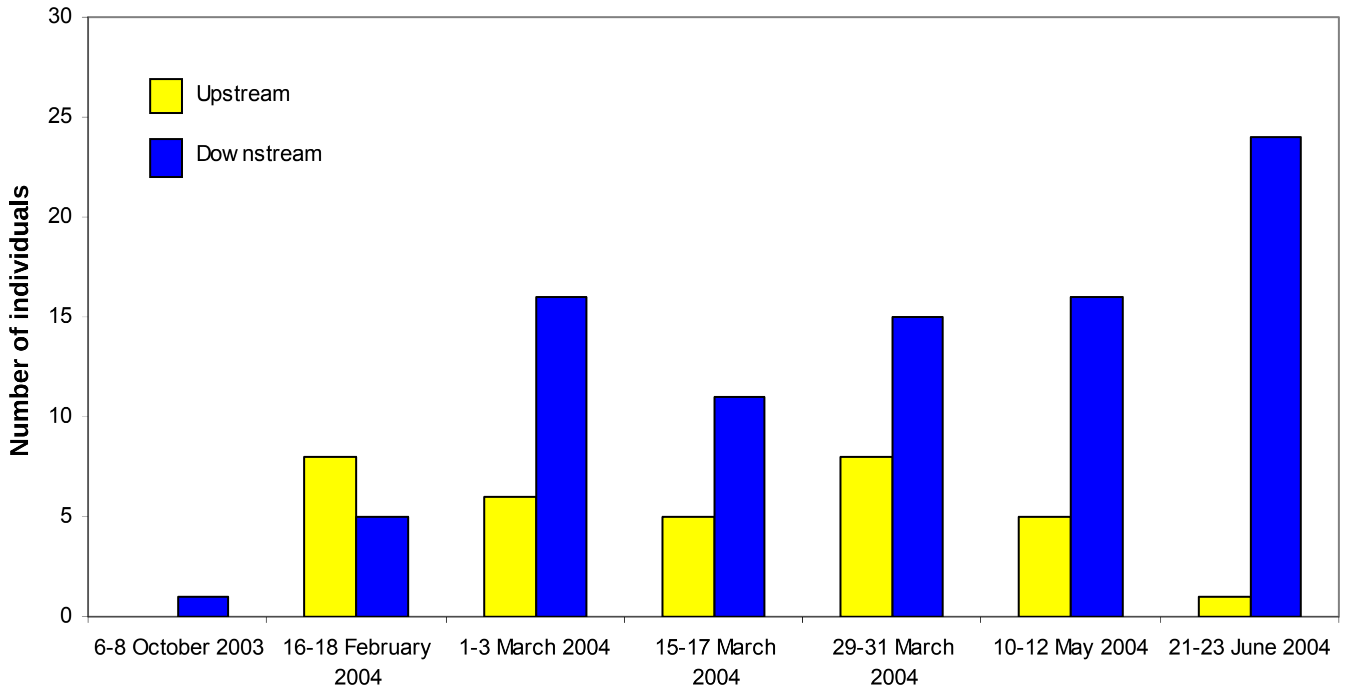

Figure 4.1.3.11. Differences in Hydracarina numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

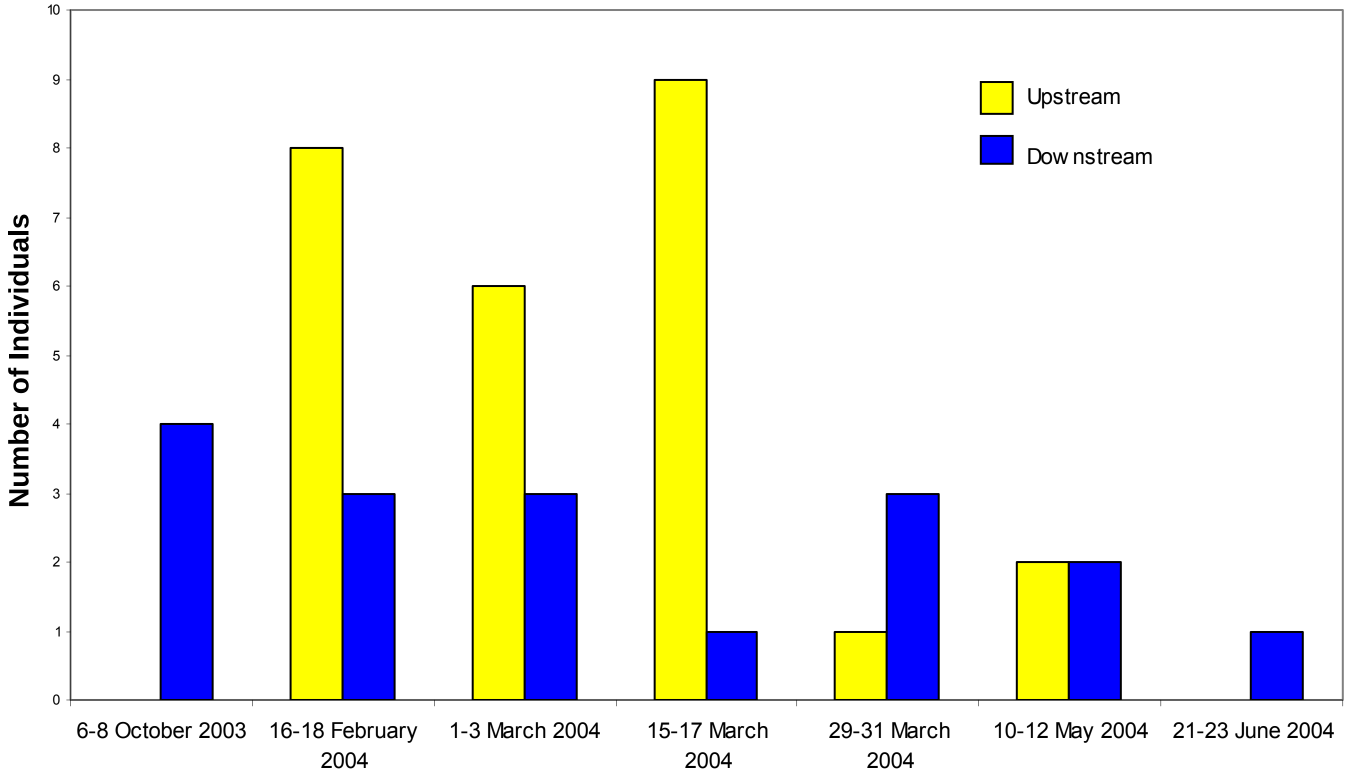

Figure 4.1.3.12. Differences in Crustacea numbers found during the sampling period, October 2003-June 2004, between sites upstream and downstream from the Klein Plaas dam, situated in the Eerste River, Jonkershoek.

4.2 SOUTH AFRICAN SCORING SYSTEM 5TH EDITION (SASS5)

An indication of the quality of each site was obtained using the SASS5 scoring system (Fig. 4.2.1). The sampling area upstream of the Klein Plaas dam scored > the downstream sampling area. The Yellow foot sampling station (downstream) had an overall mean of 46.96, ranging from 8-95. The Concrete bridge station (just upstream of the dam) had an overall mean of 91.59, ranging from 34-151. The White bridge station (furthest upstream) had the highest scores with an overall mean of 105.26, ranging from 67-160.

SASS5 values, Number of Taxa and ASPT scores recorded at each of the sample sites for each of the sample periods are given in Table 4.2.1a & b.

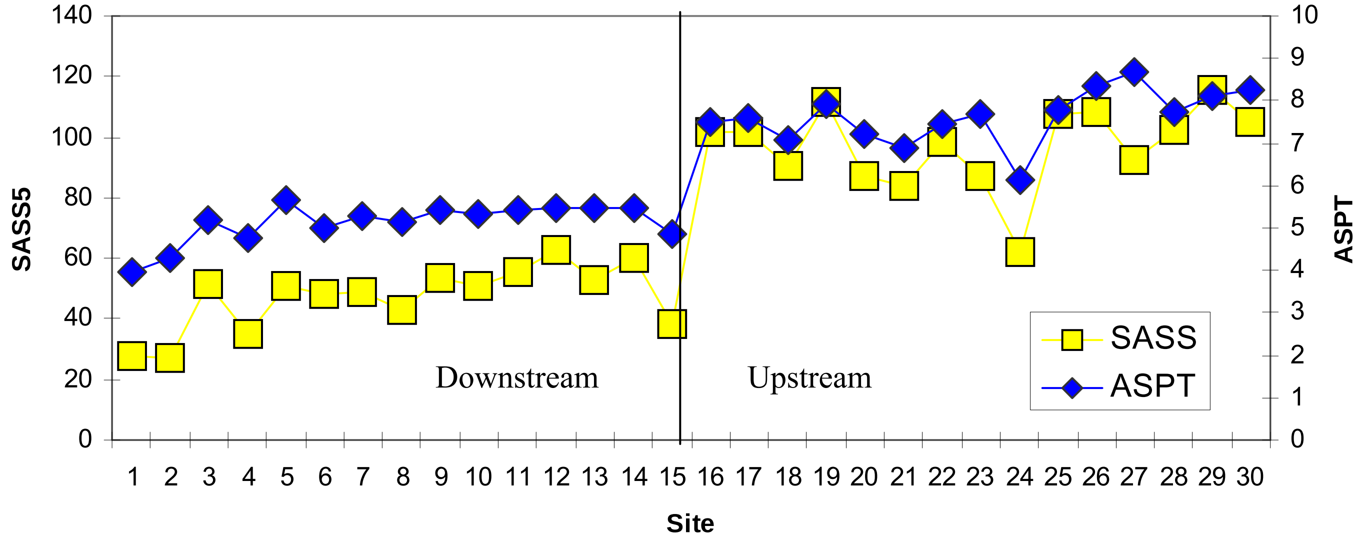

Figure 4.2.1. Values obtained from the calculation of the average SASS5 and ASPT score for each of the sampling sites along the Eerste River, Jonkershoek.

Table 4.2.1a. South African Scoring System (SASS5) values, Number (#) of Taxa and the Average Score per Taxa (ASPT) values obtained for each of the sampling sites along the Eerste River, Jonkershoek.

Site	6-8 October 2003			16-18 February 2004			1-3 March 2004			15-17 March 2004		
	SASS	# Taxa	ASPT	SASS	# Taxa	ASPT	SASS	# Taxa	ASPT	SASS	# Taxa	ASPT
1	34	8	4.25	22	5	4.40	17	5	3.40	27	8	3.38
2	35	6	5.83	27	6	4.50	25	6	4.17	18	5	3.60
3	46	8	5.75	40	8	5.00	68	13	5.23	43	9	4.78
4	18	4	4.50	22	6	3.67	18	5	3.60	28	7	4.00
5	56	9	6.22	40	7	5.71	78	13	6.00	38	7	5.43
6	34	7	4.86	62	10	6.20	71	13	5.46	53	11	4.82
7	20	5	4.00	59	9	6.56	92	15	6.13	54	10	5.40
8	43	7	6.14	48	9	5.33	37	8	4.63	53	10	5.30
9	38	8	4.75	43	8	5.38	75	12	6.25	72	13	5.54
10	8	2	4.00	47	8	5.88	90	11	8.18	33	8	4.13
11	62	11	5.64	47	8	5.88	36	8	4.50	39	9	4.33
12	55	9	6.11	47	9	5.22	83	14	5.93	55	11	5.00
13	36	7	5.14	59	10	5.90	39	8	4.88	45	8	5.63
14	13	5	2.60	95	14	6.79	68	12	5.67	47	9	5.22
15	18	6	3.00	44	7	6.29	20	5	4.00	23	6	3.83
16	87	12	7.25	151	17	8.88	104	15	6.93	106	14	7.57
17	88	10	8.80	115	14	8.21	92	13	7.08	115	14	8.21
18	87	10	8.70	67	10	6.70	108	14	7.71	50	10	5.00
19	91	10	9.10	90	11	8.18	139	18	7.72	74	12	6.17
20	93	12	7.75	92	14	6.57	74	11	6.73	69	10	6.90
21	76	9	8.44	54	8	6.75	89	13	6.85	73	13	5.62
22	106	12	8.83	79	10	7.90	95	13	7.31	84	12	7.00
23	101	12	8.42	51	7	7.29	103	13	7.92	63	8	7.88
24	134	16	8.38	54	8	6.75	54	8	6.75	48	10	4.80
25	90	11	8.18	79	13	6.08	116	14	8.29	113	14	8.07
26	99	10	9.90	113	14	8.07	84	11	7.64	95	12	7.92
27	124	13	9.54	95	11	8.64	119	15	7.93	88	11	8.00
28	110	13	8.46	101	12	8.42	111	15	7.40	99	13	7.62
29	107	12	8.92	131	17	7.71	121	15	8.07	121	15	8.07
30	147	13	11.31	102	14	7.29	67	8	8.38	76	10	7.60

Table 4.2.1b. South African Scoring System (SASS5) values, Number (#) of Taxa and the Average Score per Taxa (ASPT) values obtained for each of the sampling sites along the Eerste River, Jonkershoek.

Site	29-31 March 2004			10-12 May 2004			21-23 June 2004		
	SASS	# Taxa	ASPT	SASS	# Taxa	ASPT	SASS	# Taxa	ASPT
1	35	9	3.89	19	6	3.17	42	8	5.25
2	38	10	3.80	24	6	4.00	24	6	4.00
3	55	11	5.00	70	14	5.00	39	7	5.57
4	94	14	6.71	22	4	5.50	44	8	5.50
5	67	11	6.09	46	9	5.11	30	6	5.00
6	53	10	5.30	37	9	4.11	29	7	4.14
7	33	8	4.13	27	7	3.86	56	8	7.00
8	33	8	4.13	58	10	5.80	27	6	4.50
9	66	12	5.50	30	7	4.29	51	8	6.38
10	63	13	4.85	47	10	4.70	66	12	5.50
11	54	10	5.40	81	13	6.23	70	12	5.83
12	50	10	5.00	42	10	4.20	108	16	6.75
13	69	13	5.31	85	15	5.67	35	6	5.83
14	76	13	5.85	65	10	6.50	57	10	5.70
15	31	8	3.88	93	14	6.64	38	6	6.33
16	96	15	6.40	73	10	7.30	96	12	8.00
17	106	16	6.63	93	14	6.64	104	14	7.43
18	133	18	7.39	95	14	6.79	94	13	7.23
19	127	17	7.47	145	17	8.53	116	14	8.29
20	69	10	6.90	98	13	7.54	114	14	8.14
21	66	12	5.50	126	18	7.00	103	13	7.92
22	124	18	6.89	92	14	6.57	109	14	7.79
23	85	12	7.08	78	11	7.09	128	16	8.00
24	39	9	4.33	34	7	4.86	71	10	7.10
25	125	17	7.35	126	16	7.88	105	12	8.75
26	101	15	6.73	160	19	8.42	107	11	9.73
27	67	8	8.38	95	11	8.64	58	6	9.67
28	89	13	6.85	95	14	6.79	111	13	8.54
29	108	13	8.31	100	14	7.14	122	14	8.71
30	124	17	7.29	128	17	7.53	92	11	8.36

Single Factor ANOVA tests between the data from sampling areas upstream and downstream from the Klein Plaas dam indicated significant differences between all SASS5 and ASPT scores for each sampling period as well as for the combined data set. The number of individuals found per sample period was also significantly different for all the sample periods as well as the combined data set, except for the three sampling periods during March 2004.

In Table 4.2.2, upstream shows a higher mean for SASS5, ASPT as well as the number of individuals found per site. The differences in means for SASS5, ASPT and Number of individuals are 49.99, 2.51 and 60.97 respectively. For both SASS5 and Number of individuals the totals for upstream is almost twice the number of downstream.

The SASS5 scores for upstream had the highest mean (102.67) in the spring months, with lowest mean (84.93) in mid-March.

ASPT values showed the highest mean (8.8) in spring and the lowest value (6.9) in late-March. The lowest mean for the number of individuals per sample (81.67) upstream was during early-March, while the highest (192) was during June. Downstream SASS5 scores had the highest mean values (54.47) throughout March, with the lowest mean (34.4) during spring. The ASPT values for upstream had the highest mean (5.55) during late winter and lowest mean (4.69) during mid-March. The lowest mean for the numbers of individuals captured per sample (54.8) for upstream was during summer, while the highest (83.33) was during spring.

Table 4.2.2. Comparison of South African Scoring System (SASS5) value, Average score per Taxa (ASPT) and Number of Individuals obtained from studies in the Eerste River, Jonkershoek, between the area upstream and downstream from the Klein Plaas dam, for the sample period October 2003 - June 2004.

	6-8 October 2003					16-18 February 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
SASS	102.67	379.24	34.40	269.83	Yes	91.60	835.40	46.80	327.89	Yes
ASPT	8.80	0.90	4.85	1.29	Yes	7.56	0.73	5.51	0.74	Yes
Number of individuals	134.80	2336.17	83.33	1027.95	Yes	89.87	2474.55	54.80	312.31	Yes
	1-3 March 2004					15-17 March 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
SASS	98.40	506.97	54.47	769.70	Yes	84.93	529.07	41.87	210.98	Yes
ASPT	7.51	0.31	5.20	1.54	Yes	7.09	1.34	4.69	0.57	Yes
Number of individuals	81.67	2295.10	68.47	1696.84	No	98.73	2531.64	62.40	466.83	No
	29-31 March 2004					10-12 May 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
SASS	97.27	776.64	54.47	338.98	Yes	102.53	958.98	49.73	595.07	Yes
ASPT	6.90	1.00	4.99	0.78	Yes	7.25	0.90	4.98	1.11	Yes
Number of individuals	118.00	4293.86	72.80	1007.60	No	190.33	8077.95	76.93	3076.35	Yes
	21-23 June 2004					All data				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
SASS	102.00	334.43	47.73	476.07	Yes	97.06	619.79	47.07	446.24	Yes
ASPT	8.24	0.59	5.55	0.78	Yes	7.62	1.17	5.11	1.01	Yes
Number of individuals	192.00	8934.71	59.87	1588.55	Yes	129.34	5978.88	68.37	1323.12	Yes

4.2.1 Sample period: 6-8 October 2003

There was a significant difference between the sites from upstream compared to those from downstream during this sampling period. The two sampling areas upstream showed little variation. SASS5 scores showed a curve with four peaks and four lows. The peaks were at Sites 2, 5, 8, and 11, with the latter being the highest (62). The lows were located at Sites 4, 7, 10 and 14, with the lowest at Site 10 (8). ASPT has a similar curve, except that the first peak was at Site 2, the highest value (6.22) at Site 5 and the lowest value (2.6) at Site 14. At the Concrete bridge station, SASS5 values showed a steady increase that suddenly dropped to the lowest value of 76 (Site 21), then recovered and peaked at a value of 134 (Site 24). ASPT had a more rapid climb towards the highest value of 9.1 (Site 19) from where it dropped to the lowest value of 7.75 (Site 20). SASS5 scores for the White bridge station showed two peaks, the first at Site 26 and the other at Site 30. Site 30 had the highest value (147), while the lowest value (90) was at Site 25. ASPT showed the same trend, except that the first peak was at Site 27. The highest value was 11.31 (Site 30), and lowest 8.14 (Site 25).

4.2.2 Sample period: 16-18 February 2004

For the second sampling period, 16-18 February 2004, the Yellow food sampling area's values were slightly higher than in the previous sampling period, with SASS5 scores showing a steady increase, with peaks at Sites, 3, 6 and 14. The highest value (95) was located at Site 14 and the lowest (22) at Site 4. ASPT scores showed a similar trend, except peaking at Site 7 instead of 6. The highest value (6.79) and lowest value (3.67) were at Sites 4 and 14 respectively. Concrete bridge station showed declining abundance instead of the increase of the previous sampling period. SASS5 scores were highest (151) at Site 16 and dropped to the lowest at Site 23. Along this decline the curve peaks twice at Sites 20 and 22. ASPT curve also starts at its highest point (8.88) at Site 16, from where follows the same trend as the SASS5 curve, except for having the lowest value (6.57) at Site 20, and peaking at Site 19 instead of 20. Furthermore, there is a decline between Sites 23 and 24 instead of an increase. White bridge station ASPT curve is bell-shaped and starts at the lowest value (6.08) at Site 25 and peaks at Site 27 with a value of 8.64. SASS5 scores also start at the lowest value of 79 (Site 25), from where the curve peaks twice at 26 and 29 (the last is the highest with value of 131).

4.2.3 Sample period: 1-3 March 2004

Early-March data were very variable, with the SASS5 and ASPT scores varying greatly between sites. SASS5 and ASPT scores for Yellow foot station had a curve with six peaks at Sites 3, 5, 7, 10, 12 and 14, with the maximum SASS5 value of 92 (Site 7), ASPT of 8.18 (Site 10) and minimum SASS5 value of 17 (Site 1) and ASPT of 3.4 (Site 1). Concrete bridge station showed a decreasing curve that peaked twice at Site 19 (highest value of 139) and 23 from where it dropped to its lowest value of 54 (Site 24). ASPT showed the same trend with exception that there was no overall decline. The minimal value for this sampling area was 6.73 (Site 20) and maximum value 7.92 (Site 23). SASS5 scores for White bridge station indicated an variable curve with three peaks at Sites 15, 17 and 19, with the maximum value of 121 (Site 29) and minimum value of 67 (Site 30). ASPT showed the same trend, but with less variation for consecutive sites, also the last transition from Sites 29 to 30 increased instead of declining, giving the lowest value (7.4) at Site 28 and highest (8.38) at Site 30.

4.2.4 Sample period: 15-17 March 2004

For mid-March, the data were less variable than in the previous sample period downstream, but more variable for upstream. Upstream still gave higher values than downstream. SASS5 values for Yellow foot station gave a curve with five peaks located at Sites 1, 3, 9, 12 and 14, similar to the curve for October in the same sampling area. For this sampling area, the maximum value was 72 (Site 9) and minimum 18 (Site 2). ASPT scores showed a similar trend, with peaks at Sites 3, 5, 7, 9 and 13, giving the maximum of 5.63 (Site 13) and minimum of 3.38 (Site 1). The curve for Concrete bridge station was close to the one in February. It starts high and declines, with three peaks. For SASS5 values, the peaks lie at Sites 17, 19 and 22, with maximum value (115) at Site 17 and minimum (48) at Site 24. ASPT peaks were at Sites 17, 20 and 23, with maximum value (8.21) at Site 17 and minimum value (4.8) at Site 24. For White bridge station, SASS5 scores agree with the February sample period, while ASPT scores compare with to mid-March sample period. SASS5 scores were also much more variable than ASPT scores. SASS5 values start as a peak at Site 25, dropping to a low at Site 26, climbing to the highest point (121) at Site 29, from where it drops to the lowest value (76) at Site 30. ASPT was almost a straight line, with only two dips, at Site 28 and 30. The minimum value for this curve was 7.6 (Site 30) and the maximum was 8.07 (Site 25 and 29).

4.2.5 Sample period: 29-31 March 2004

SASS5 scores for Yellow foot station, showed a curve with three peaks at Sites 4, 9 and 14, with a maximum of 94 (Site 4) and a minimum of 31 (Site 15). ASPT curve showed the same trend, with only an additional peak at Site 11. The maximum value was 6.71 (Site 4) and the minimum value was 3.88 (Site 15). The SASS5 scores for Concrete bridge station showed two very high peaks at Sites 18 and 22, and two very low troughs at Sites 21 and 24. The maximum value for this sampling area was 133 (Site 18) and the minimum 39 (Site 24). ASPT showed the same trend, with the two peaks at Sites 19 and 23. Maximum ASPT value was 7.47 (Site 19) and minimum 4.33 (Site 24). White bridge station showed an increase in variance between sites. SASS5 value showed a v-shaped curve with peaks at Sites 25 and 30, and minimum (67) at Site 27. ASPT scores showed a zigzag curve as in the previous two sampling periods, but with an increase in variance between the points. Maximum value for the sampling area was 8.38 (Site 27), and minimum 6.73 (Site 26).

4.2.6 Sample period: 10-12 May 2004

There was little difference between the sampling areas upstream and downstream. SASS5 scores for Yellow foot station showed a zigzag curve with an overall increase, starting at the minimum of 19 (Site 1) and ending at the maximum of 93 (Site 15). In between, there were three peaks, at Sites 4, 8 and 11, with the trough at Sites 7, 9 and 12. ASPT scores were similar, with a minimum of 3.17 (Site 1) and a maximum of 6.64 (Site 15), with more peaks at Sites 3, 5, 8, 11, 13 and 15. SASS5 scores for Concrete bridge station showed a curve with two peaks at Sites 19 and 21, with lows at Sites 16, 20 and 24. ASPT scores indicate a curve with three peaks at Sites 15, 19 and 23, and lows at Sites 17, 22 and 24. SASS5 scores for the White bridge station indicated a v-shaped curve with highest peak (160) at Site 26 and another at Site 30, with minimum (95) at Site 27. ASPT scores showed a curve with two peaks at Site 27 and Site 30, with a minimum (6.79) at Site 28.

4.2.7 Sample period: 21-23 June 2004

Yellow foot station showed a similar trend for both SASS5 and ASPT, curves with five peaks at Sites 1, 4, 7, 12 and 14. The minimum of 24 for SASS5 and 4.0 for ASPT, were both at Site 2. ASPT curve deviated from the trend, peaking at Site 3 instead of 4 and having an additional peak at Site 9 and one peak less at Site 14. SASS5 values for Concrete bridge station were higher than at the White bridge station. This trend is not seen in ASPT scores. SASS5 scores indicated a curve with three peaks at Sites 17,19 and 23, with a maximum of 128 (Site 23) and a minimum of 71 (Site 24). ASPT scores also showed a less variant curve with three peaks at Sites 16, 19 and 23, with a maximum value of 8.29 (Site 19) and a minimum of 7.1 (Site 24). For the White bridge station, the SASS5 scores indicated a curve with two peaks at Sites 26 and 29. The curve presents a maximum of 122 (Site 29) and minimum of 58 (Site 27). Furthermore, ASPT values showed a less variable curve peaking at Site 26 with a maximum value of 9.73, from where the curve declines to a minimum of 8.36 (Site 30).

4.3 ABIOTIC VARIABLES

4.3.1 Temperature

The sampling area maintained a mean water temperature of 16.4 ^{o}C over the entire sampling period, with a maximum of 22.9 ^{o}C in February and a minimum of 13.2 ^{o}C in late-March.

Downstream, during the whole study period, there was a higher mean water temperature than upstream, with the greatest difference in the early March (6.63 ^{o}C) and smallest difference in June (0.34 ^{o}C). The combined data set indicated a 2.63 ^{o}C difference in means (Table 4.3.1). The mean water temperature for each of the sampling sites is shown in Fig. 4.3.1.

Single factor ANOVA test indicated significant differences in temperature between the sampling areas upstream and downstream, for all of the sampling periods as well as the combined data set.

4.3.2 Conductivity

The mean conductivity was 0.05 mSm^{-1} during the whole sampling period, with a maximum of 0.076 mSm^{-1} in early and mid-March and a minimum of 0.023 mSm^{-1} in June. The mean values (Table 4.3.1) are constantly higher downstream, with greatest difference in March (0.04 mSm^{-1}) and smallest difference (0.02 mSm^{-1}) in October, February, May and June. The overall data set also has a difference in mean conductivity of 0.02 mSm^{-1}.

Table 4.3.1. Comparison of physical and chemical data obtained in the Eerste River, Jonkershoek, between the areas upstream and downstream of the Klein Plaas dam. Sample period October 2003 - June 2004.

	6-8 October 2003					16-18 February 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
Temperature	14.55	0.18	15.38	0.33	Yes	19.34	0.19	22.06	0.30	Yes
Conductivity	0.03	0.00	0.05	0.00	Yes	0.05	0.00	0.07	0.00	Yes
% Dissolved Oxigen	102.46	1.46	102.20	8.21	No	44.87	108.77	39.57	13.92	No
Dissolved Oxygen mg/l	10.40	0.03	10.14	0.06	Yes	4.14	0.96	3.47	0.11	Yes
pH	5.56	0.03	6.73	0.16	Yes	6.08	0.03	6.85	0.01	Yes
	1-3 March 2004					15-17 March 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
Temperature	14.99	0.70	21.62	0.21	Yes	17.32	0.33	19.11	0.12	Yes
Conductivity	0.04	0.00	0.08	0.00	Yes	0.05	0.00	0.08	0.00	Yes
% Dissolved Oxigen	41.50	37.75	32.29	64.66	Yes	32.97	54.64	29.84	17.19	No
Dissolved Oxygen mg/l	4.20	0.42	2.84	0.49	Yes	3.17	0.54	2.77	0.14	No
pH	5.44	0.54	6.87	0.02	Yes	6.17	0.09	6.65	0.04	Yes
	29-31 March 2004					10-12 May 2004				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
Temperature	13.81	0.24	18.49	0.05	Yes	14.44	0.17	15.84	0.01	Yes
Conductivity	0.03	0.00	0.07	0.00	Yes	0.04	0.00	0.06	0.00	Yes
% Dissolved Oxigen	35.51	38.50	32.29	39.80	No	40.00	56.52	48.68	61.55	Yes
Dissolved Oxygen mg/l	3.69	0.45	4.06	13.41	No	4.12	0.62	4.63	0.64	No
pH	6.37	0.02	6.74	0.01	Yes	6.10	0.06	6.96	0.13	Yes
	21-23 June 2004					All data				
	Up stream area		Down stream area			Up stream area		Down stream area		
	Mean	Variance	Mean	Variance	Sign.	Mean	Variance	Mean	Variance	Sign.
Temperature	11.05	0.02	11.39	0.09	Yes	15.07	6.26	17.70	12.51	Yes
Conductivity	0.03	0.00	0.05	0.00	Yes	0.04	0.00	0.06	0.00	Yes
% Dissolved Oxigen	61.73	232.30	44.67	81.11	Yes	51.29	587.17	47.08	592.06	No
Dissolved Oxygen mg/l	6.80	2.78	4.93	0.90	Yes	5.22	6.44	4.69	7.71	No
pH	4.95	0.28	7.22	1.32	Yes	5.81	0.36	6.86	0.26	Yes

The mean conductivity for each of the sampling sites is showed in Fig. 4.3.2. Single factor ANOVA test indicates significant differences in conductivity between upstream and downstream for each of the sampling periods, as well as for the combined data set.

4.3.3 Dissolved oxygen

The mean percentage dissolved oxygen for the whole sampling period was 49.2% and mean dissolved oxygen was 4.88 mg.ℓ^{-1}. For the combined data set, the maximum were 106.1% and 10.64 mg/ℓ, both in late-March, and the minimum 18.3% and 1.72 mg.ℓ^{-1}, both in October.

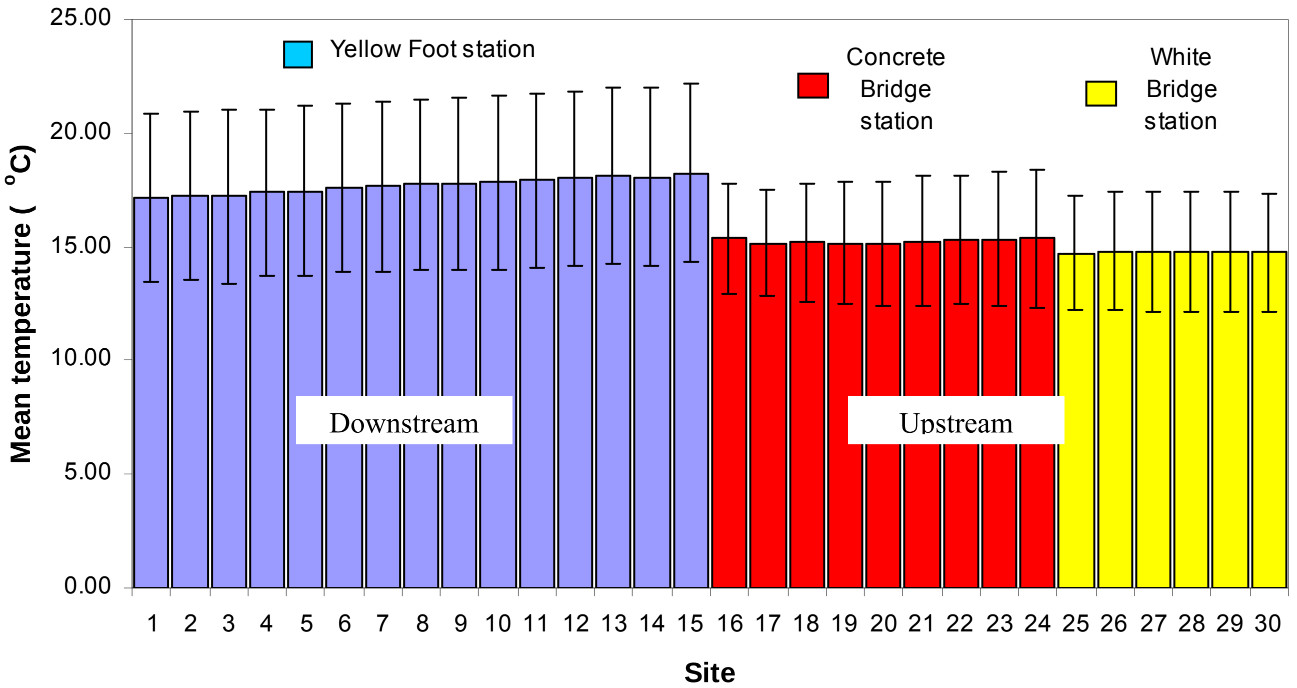

Figure 4.3.1. Mean Temperature in °C per site, measured in the Eerste River catchment area, during the whole sampling period.

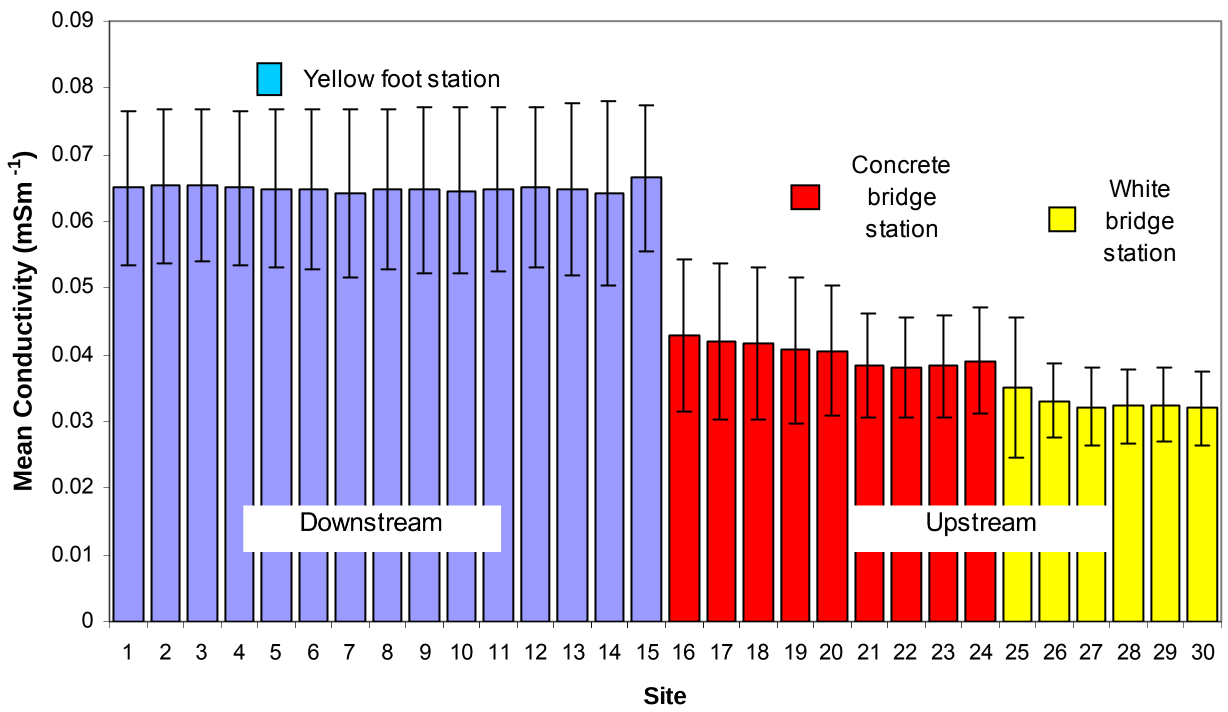

Figure 4.3.2. Mean Conductivity in mSm^{-1} per site, measured in the Eerste River catchment area, during the whole sampling period.

For both percentage dissolved oxygen and dissolved oxygen in mg.ℓ$^{-1}$ (Table 4.3.1), upstream means were higher, except for late-March, where the dissolved oxygen in mg.ℓ$^{-1}$, and in May, where both percentage and dissolved oxygen in mg.ℓ$^{-1}$ were greatest downstream. June experienced the greatest difference between means for percentage dissolved oxygen (17.06 %) and for dissolved oxygen in mg.ℓ$^{-1}$ (1.87 mg.ℓ$^{-1}$), while October showed the smallest difference for percentage dissolved oxygen (0.26%) and for dissolved oxygen in mg/ℓ (0.26 mg.ℓ$^{-1}$). The mean dissolved oxygen and mean percentage dissolved oxygen for each of the sampling sites are shown in Fig. 4.3.3.1 and Fig. 4.3.3.2 respectively.

Single factor ANOVA test indicate significant differences in the percentage dissolved oxygen between upstream and downstream during October, February, early-March and June. Early-March, May and June shows significant difference for the dissolved oxygen in mg.ℓ$^{-1}$. The combined data set shows no significant differences between upstream and downstream.

4.3.4 Acidity level

The sampling area had a mean pH value of 6.34 over the whole sampling period, with the most alkaline site (pH=10.28) in June and the most acidic site (pH=4.19) in early-March.

Upstream sites were more acidic than downstream (Table 4.3.1), with the greatest difference in means in June (2.27) and the weakest in late-March (0.37). The overall data set calculated a difference in pH of 1.05 between upstream and downstream. The mean pH values for each of the sampling sites are shown in Fig. 4.3.4.

Single factor ANOVA test indicated significant differences in pH values between upstream and downstream for each of the sampling periods, as well as for the combine data set.

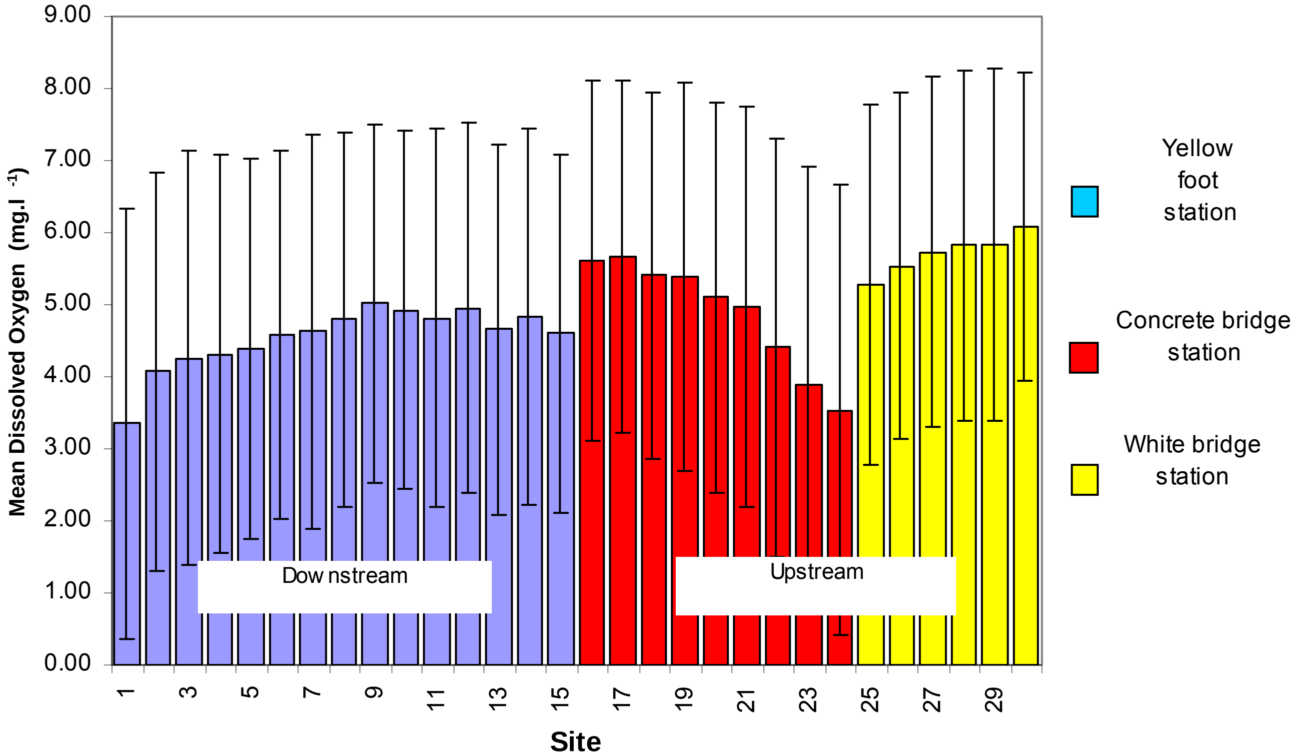

Figure 4.3.3.1. Mean Dissolved Oxygen in mg.l^{-1} per site, measured in the Eerste River catchment area, during the whole sampling period.

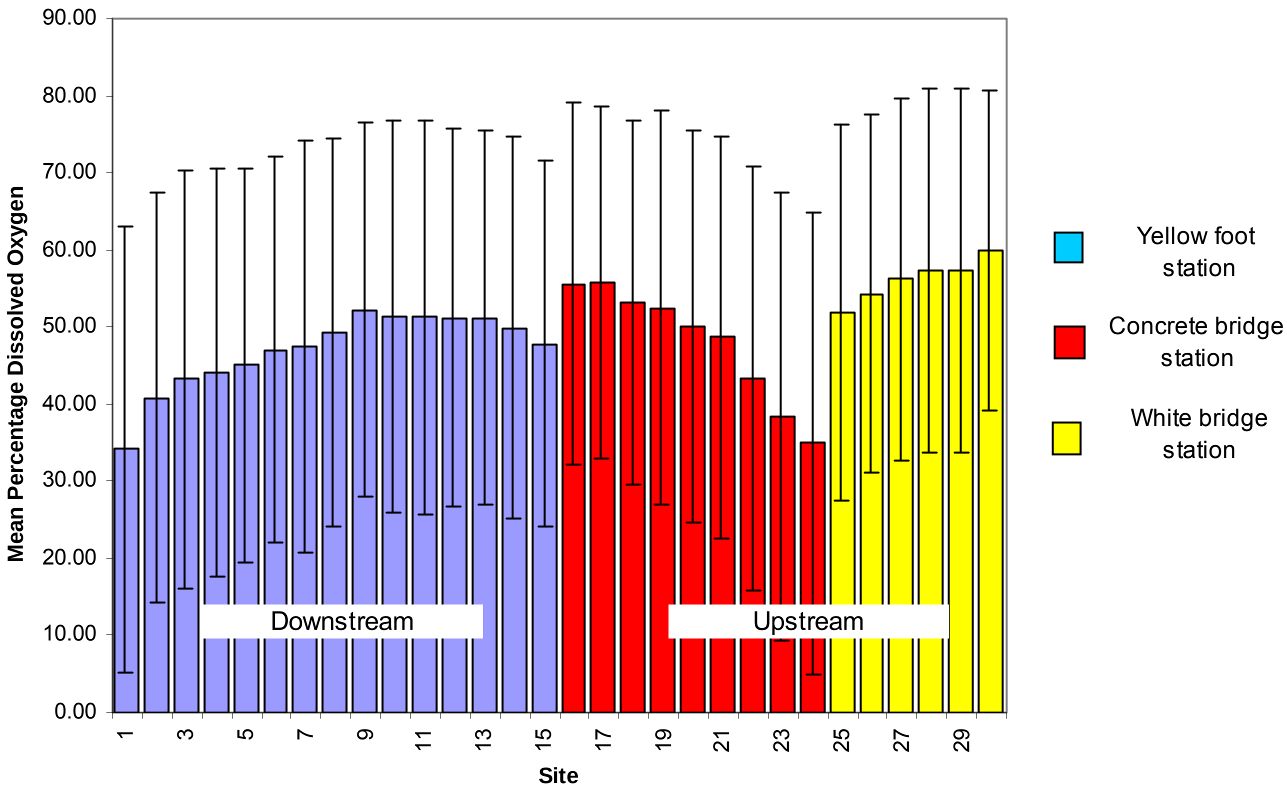

Figure 4.3.3.2. Mean Percentage Dissolved Oxygen per site, measured in the Eerste River catchment area, during the whole sampling period.

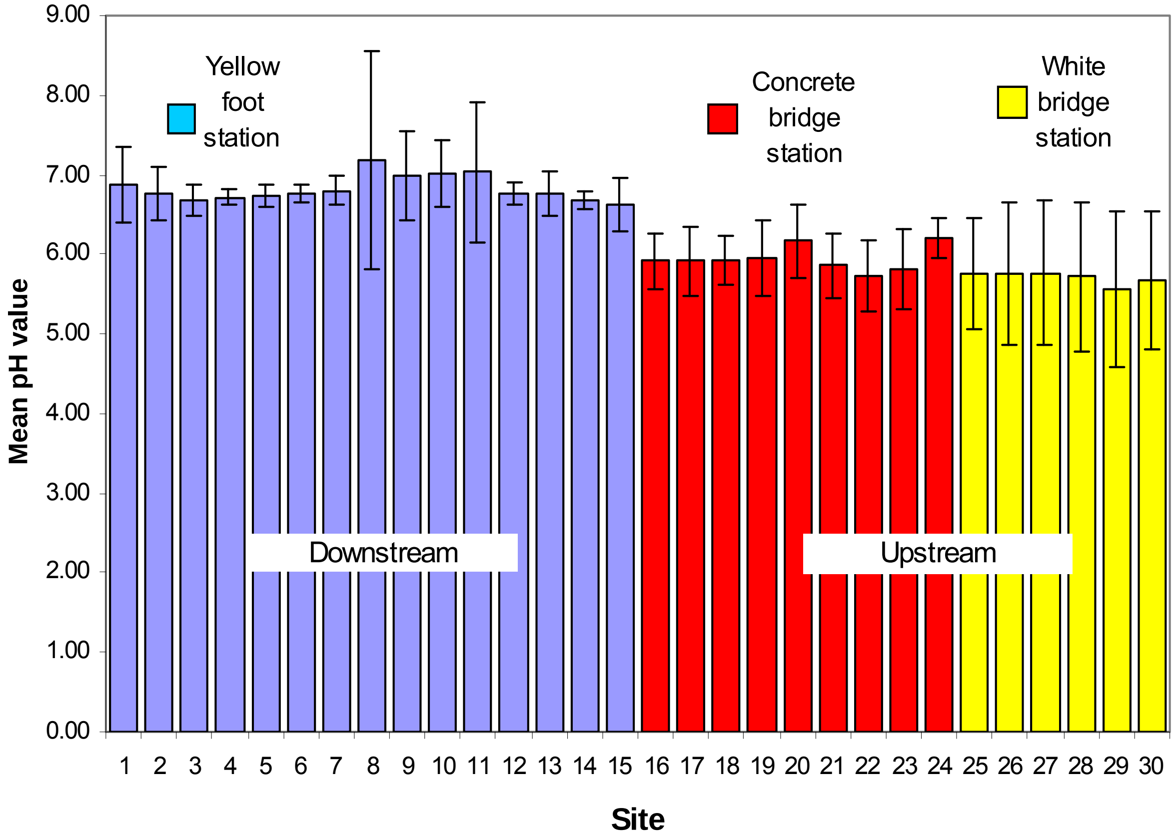

Figure 4.3.4. Mean pH per site, measured in the Eerste River catchment area, during the whole sampling period.

4.4 BRAY-CURTIS SIMILARITY

4.4.1 **Combined data set**

There were four main clusters, two each for the upstream and downstream data (Fig.4.4.1). The White bridge sites cluster together with Sites 21, 22 and 23 (73.4%), from where it joins the cluster (74.1%) of the remaining upstream sites, as well as one of the downstream clusters (75.1%). The other downstream cluster (63.1%) joins the rest at a similarity of 48.8%

4.4.2 Sampling period: 6-8 October 2003

The overall data set (Fig. 4.4.2) had a similarity of 31.5%, with the downstream sites clustering together with a similarity of 46.6%. Upstream had a similarity of 59.5% and within this grouping, Sites 24 and 21 had a similarity of 71.6% with the main group. White bridge station shows a small grouping (similarity of 87.0%), with the exception of Sites 25 and 26 joined up elsewhere with sites from the Concrete bridge station.

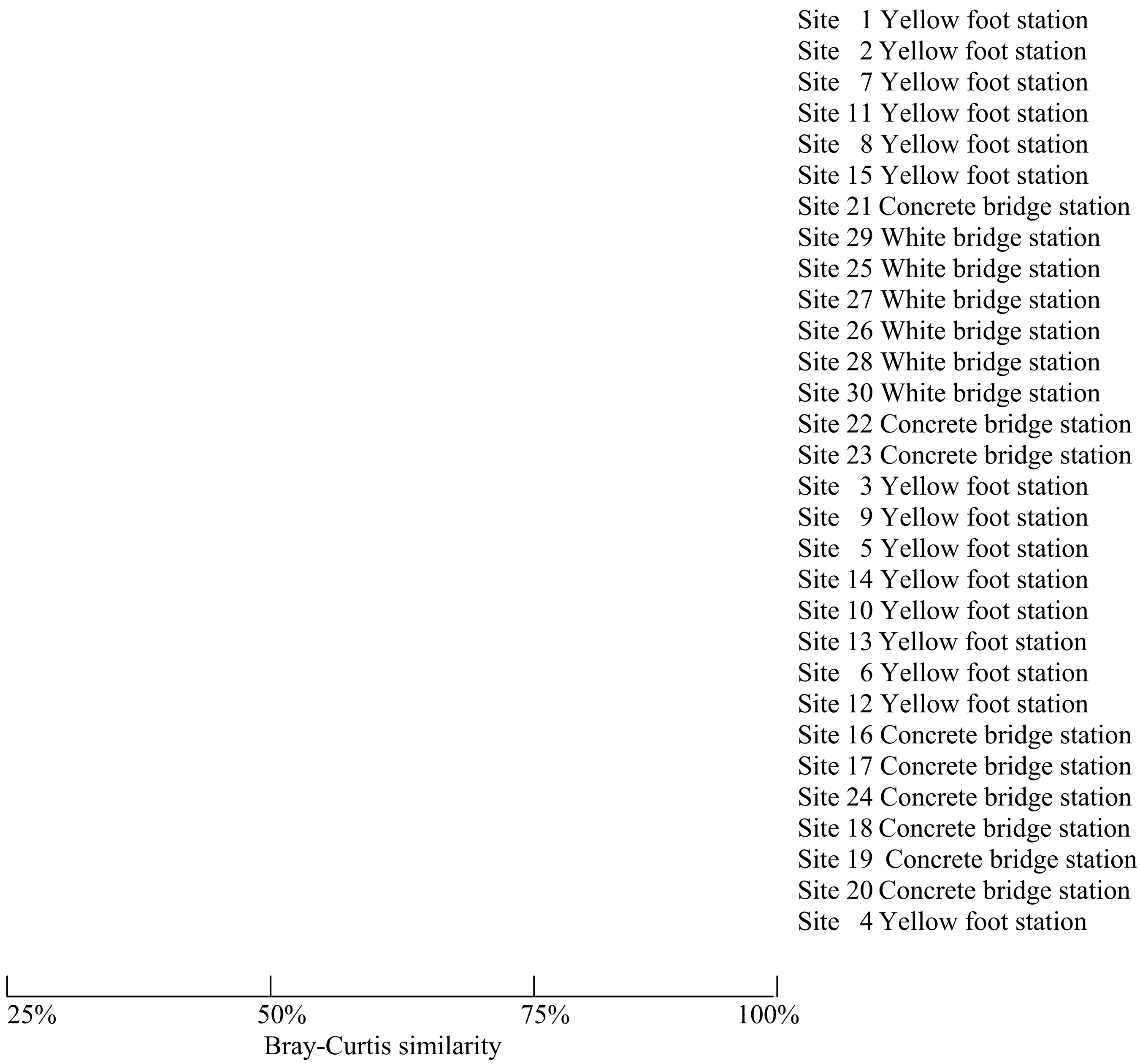

Figure 4.4.1. Dendogram of macroinvertebrate families across all 30 sites for the whole sampling period.

Site 1 Yellow foot station
Site 2 Yellow foot station
Site 8 Yellow foot station
Site 9 Yellow foot station
Site 11 Yellow foot station
Site 12 Yellow foot station
Site 13 Yellow foot station
Site 3 Yellow foot station
Site 5 Yellow foot station
Site 6 Yellow foot station
Site 14 Yellow foot station
Site 15 Yellow foot station
Site 7 Yellow foot station
Site 4 Yellow foot station
Site 10 Yellow foot station
Site 16 Concrete bridge station
Site 17 Concrete bridge station
Site 18 Concrete bridge station
Site 20 Concrete bridge station
Site 23 Concrete bridge station
Site 25 White bridge station
Site 19 Concrete bridge station
Site 22 Concrete bridge station
Site 26 White bridge station
Site 27 White bridge station
Site 29 White bridge station
Site 30 White bridge station
Site 28 White bridge station
Site 21 Concrete bridge station
Site 24 Concrete bridge station

25% 50% 75% 100%
Bray-Curtis similarity

Figure 4.4.2. Dendogram of macroinvertebrate families across all 30 sites for the October sampling period.

4.4.3 Sampling period: 16-18 February 2004

The overall data set (Fig. 4.4.3) had a similarity of 33.48%. There was no clear clustering for the different sampling areas, but the cluster with a similarity of 41% consisted mainly of sites from Yellow foot station, with some sites from Concrete bridge station (Sites 20 - 24). The cluster with a similarity of 47,37% consist mainly of all the sites from the White bridge station, the remainder of the sites from Concrete bridge station and Site 10. Sites 16 and 19 had a 100% similarity.

4.4.4 Sampling period: 1-3 March 2004

The overall data set (Fig. 4.4.4) had a similarity of 34.97%. The first cluster consisted of mainly sites from White bridge station plus some from Concrete bridge station (Sites 16-19) and had a similarity of 58.79%. From where it joins with a cluster, consisting of sites from Yellow foot station plus Sites 22-24, at a similarity of 39.91%. From here it joins up with a cluster (similarity of 42.36%), consisting of the remainder of the sites from Yellow foot station and Sites 20 and 21.

4.4.5 Sampling period: 16-18 March 2004

The overall data (Fig. 4.4.5) had a similarity of 38.87%. The first cluster consisted of the downstream sites with a similarity of 53.55%. The sites from White bridge station cluster with Site 19 having a similarity of 63.61%, from where it joins a cluster consisting of Sites 17, 21, 22 and 23 at a similarity of 59.17%. This group joins then a cluster consisting of the remainder of the sites from Concrete bridge station and Sites 1 and 4, at a similarity of 46.29%.

4.4.6 Sampling period: 29-31 March 2004

The overall data set (Fig. 4.4.6) had a similarity of 39.71%. Downstream sites all clustered together with Site 24 at a similarity of 49.78%, while the rest of the sites from upstream clustered together at a similarity of 45.18%. White bridge station clustered together with Site 19 at a similarity of 74.48%.

4.4.7 Sampling period: 10-12 May 2004

The overall data set (Fig. 4.4.7) had a similarity of 41.54%. Upstream formed a cluster with a similarity of 67.48% before joining a cluster of sites from Yellow foot station at a similarity of 57.3%. The rest of the downstream sites clusters together at a similarity of 44.08%.

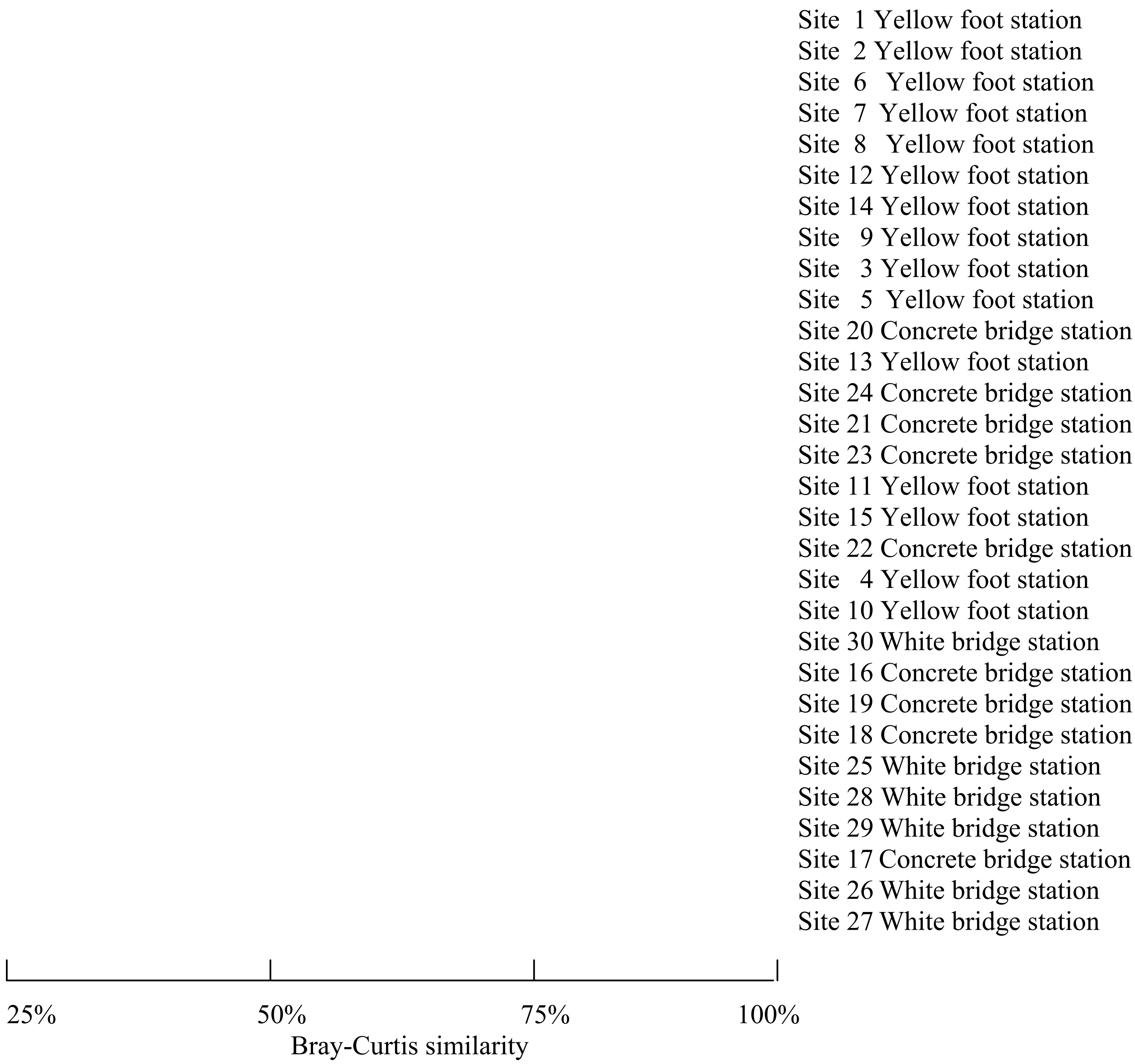

Figure 4.4.3. Dendogram of macroinvertebrate families across all 30 sites for the February sampling period.

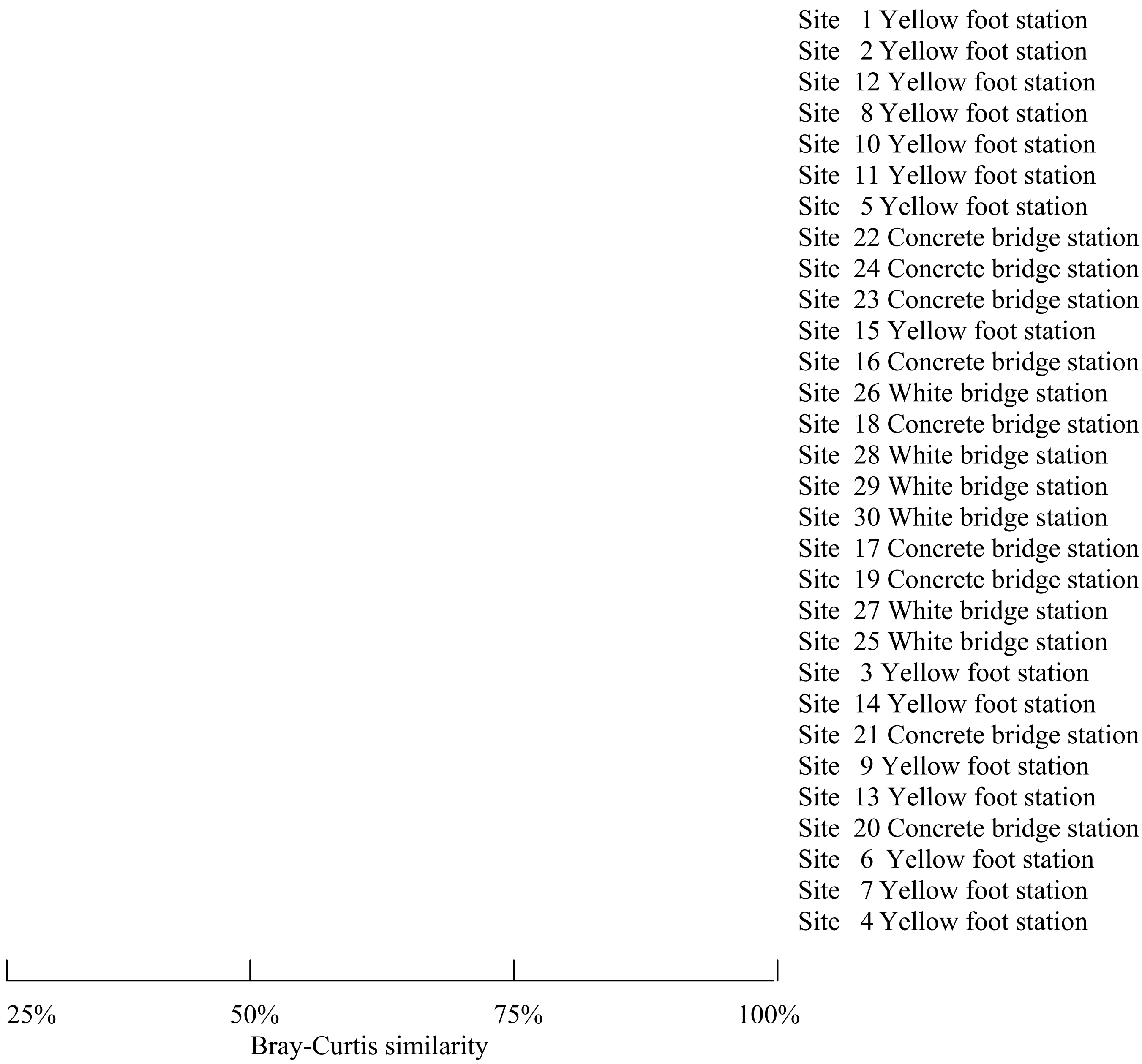

Figure 4.4.4. Dendogram of macroinvertebrate families across all 30 sites for the early-March sampling period.

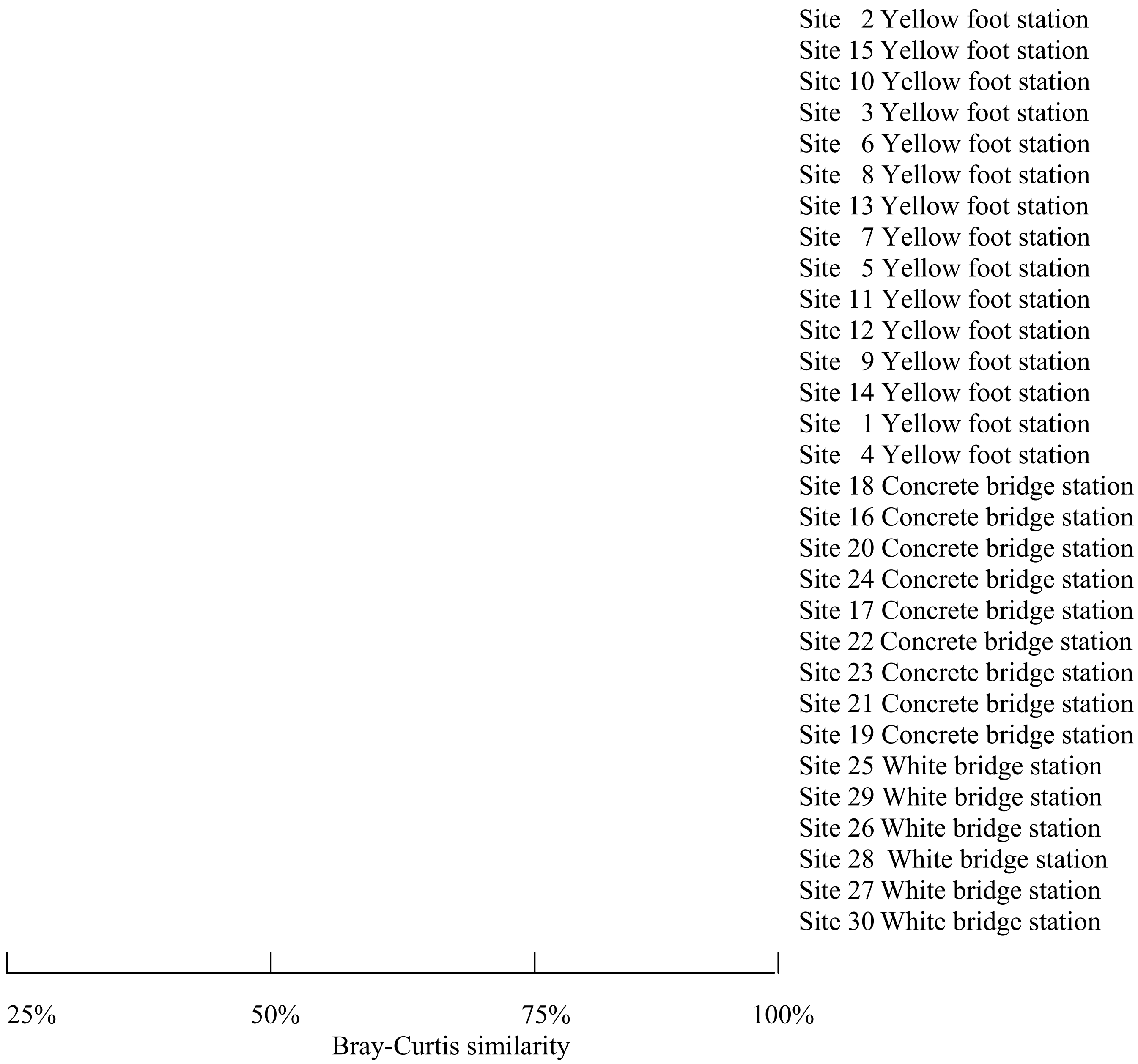

Figure 4.4.5. Dendogram of macroinvertebrate families across all 30 sites for the mid-March sampling period.

Site 1 Yellow foot station
Site 7 Yellow foot station
Site 2 Yellow foot station
Site 24 Concrete bridge station
Site 8 Yellow foot station
Site 15 Yellow foot station
Site 3 Yellow foot station
Site 5 Yellow foot station
Site 6 Yellow foot station
Site 4 Yellow foot station
Site 9 Yellow foot station
Site 10 Yellow foot station
Site 14 Yellow foot station
Site 11 Yellow foot station
Site 12 Yellow foot station
Site 13 Yellow foot station
Site 16 Concrete bridge station
Site 18 Concrete bridge station
Site 17 Concrete bridge station
Site 22 Concrete bridge station
Site 19 Concrete bridge station
Site 30 White bridge station
Site 26 White bridge station
Site 25 White bridge station
Site 28 White bridge station
Site 29 White bridge station
Site 23 Concrete bridge station
Site 20 Concrete bridge station
Site 27 White bridge station
Site 21 Concrete bridge station

25% 50% 75% 100%

Bray-Curtis similarity

Figure 4.4.6. Dendogram of macroinvertebrate families across all 30 sites for the late-March sampling period.

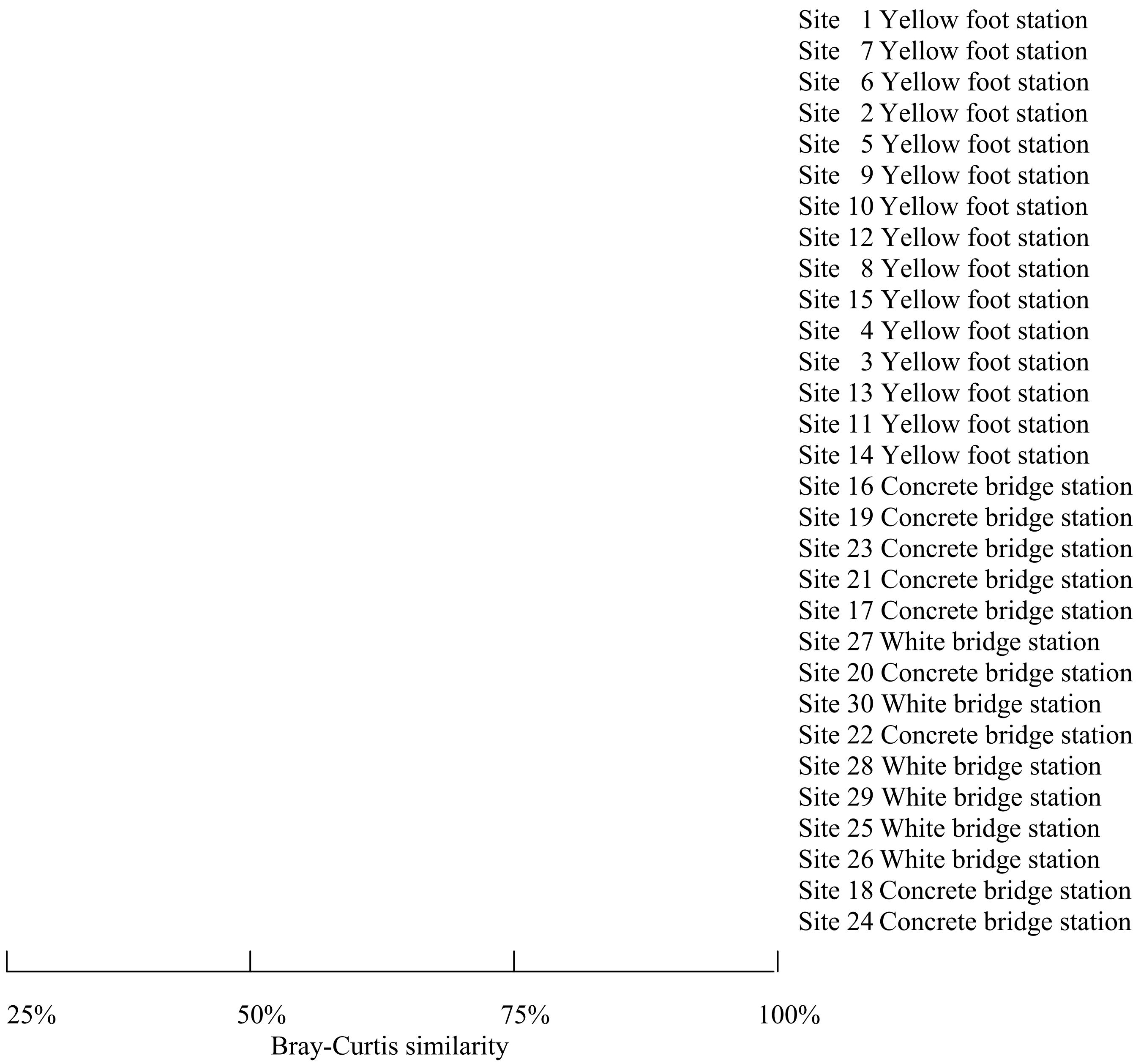

Figure 4.4.7. Dendogram of macroinvertebrate families across all 30 sites for the May sampling period.

4.4.8 **Sampling period: 21-23 June 2004**

The overall data set (Fig. 4.4.8) have a similarity of 41.25%. Upstream, with the exception of Site 27, cluster together with a similarity of 69.94%, from where it joins up at a similarity of 54.14% with a group of Yellow bridge sites. From here it joins with the combination of Sites 7 and 13. The pair of Sites 1 and 27 joins the cluster at a similarity of 45.01%, before finally joining up with a cluster of the remaining Yellow foot sites.

4.4.9 Physical and Chemical data

Downstream sites, with the exception of Site 1, cluster together with a similarity of 84.65%. The upstream sites, with the exception of Site 24 cluster together with a similarity of 76.41%. This cluster joins up with the combination of Site 1 and 24 at a similarity of 75.19% before joining the downstream cluster, to give an overall similarity for the data set of 71.46% (Fig. 4.4.9).

4.5 SIMPSON'S INDEX OF DIVERSITY

4.5.1 Combined data set

The mean Simpson's index for each of the sites (Fig. 4.5.1), shows that the Yellow foot station (Downstream) had the lowest diversity with an average mean of 4.05, the Concrete bridge and White bridge stations showed an average mean values of 6.01 and 5.7, respectively.

Site 23, with a mean of 7.3, Site 21 with a mean of 6.9 and Site 19 with a mean of 6.57, had got the highest diversities. The lowest values can be found at Sites 4 and 9, both with mean values of 3.44.

4.5.2 Sampling period: 6-8 October 2003

The Yellow foot station had a mean of 2.29 (the lowest for the whole study period), Concrete bridge station of 4.03 and White bridge station of 5.97. The most diverse site for Yellow foot station was Site 11 (3.42), for Concrete bridge, Site 17 (5.05) and Site 27 (7.3) for the White bridge station, while the minimum values were at Site 14 (1.33) for Yellow foot station, Site 16 (2.9) for Concrete bridge station and Site 25 (5.21) for White bridge station (Fig.4.5.2).

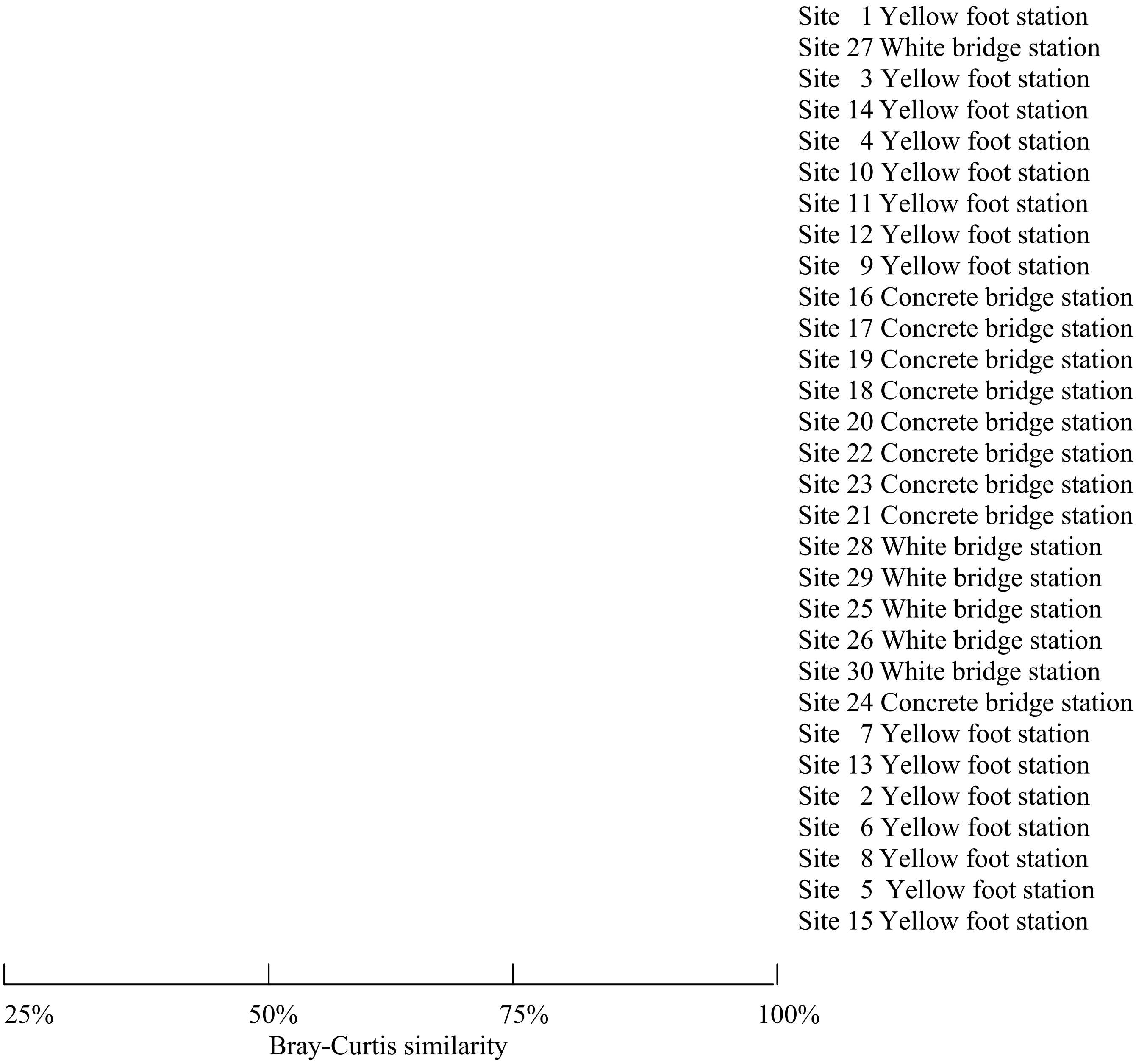

Figure 4.4.8. Dendogram of macroinvertebrate families across all 30 sites for the June sampling period.

Site 2 Yellow foot station
Site 3 Yellow foot station
Site 7 Yellow foot station
Site 15 Yellow foot station
Site 11 Yellow foot station
Site 12 Yellow foot station
Site 13 Yellow foot station
Site 4 Yellow foot station
Site 5 Yellow foot station
Site 6 Yellow foot station
Site 8 Yellow foot station
Site 10 Yellow foot station
Site 14 Yellow foot station
Site 9 Yellow foot station
Site 1 Yellow foot station
Site 24 Concrete bridge station
Site 16 Concrete bridge station
Site 26 White bridge station
Site 22 Concrete bridge station
Site 23 Concrete bridge station
Site 17 Concrete bridge station
Site 19 Concrete bridge station
Site 18 Concrete bridge station
Site 25 White bridge station
Site 21 Concrete bridge station
Site 27 White bridge station
Site 28 White bridge station
Site 29 White bridge station
Site 30 White bridge station
Site 20 Concrete bridge station

70% 80% 90% 100%

Bray-Curtis similarity

Figure 4.4.9. Dendogram of physical and chemical data of the water across all 30 sites for the whole sampling period.

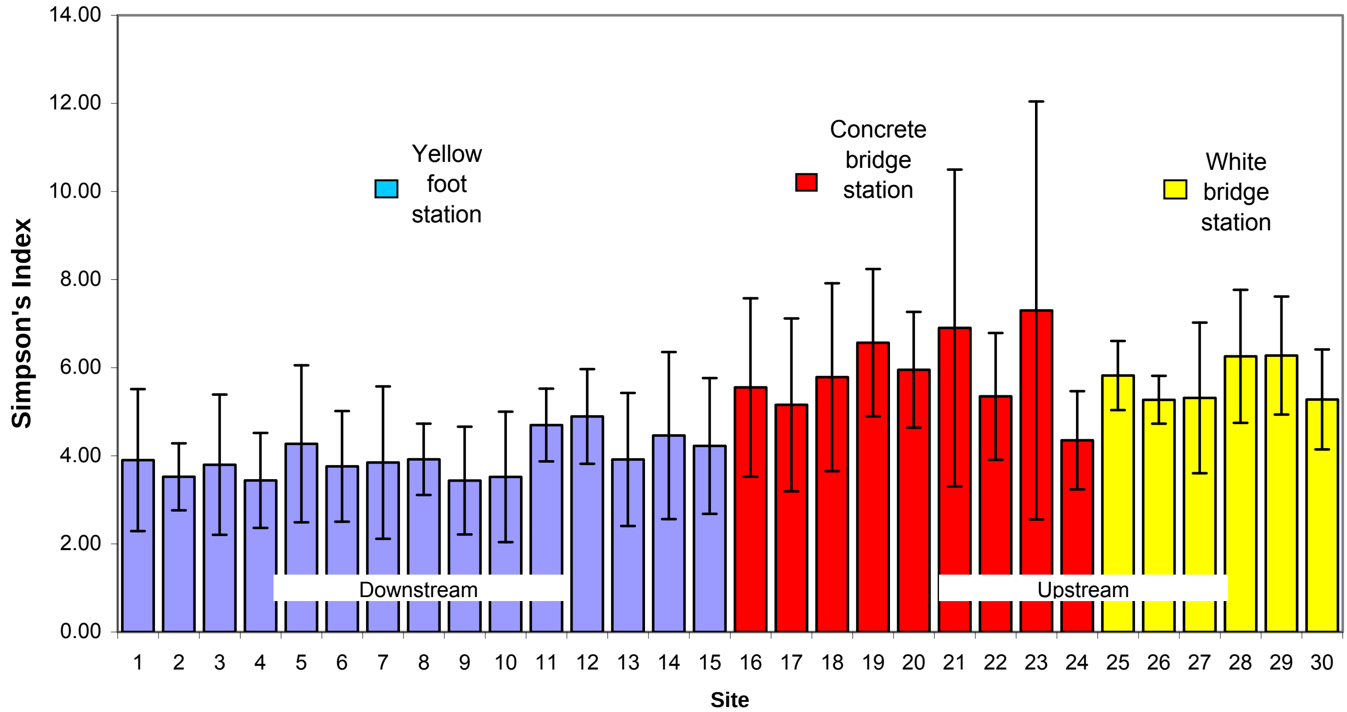

Figure 4.5.1. The Average Simpson's index values of Diversity $\{D_s=\Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, October 2003 - June 2004.

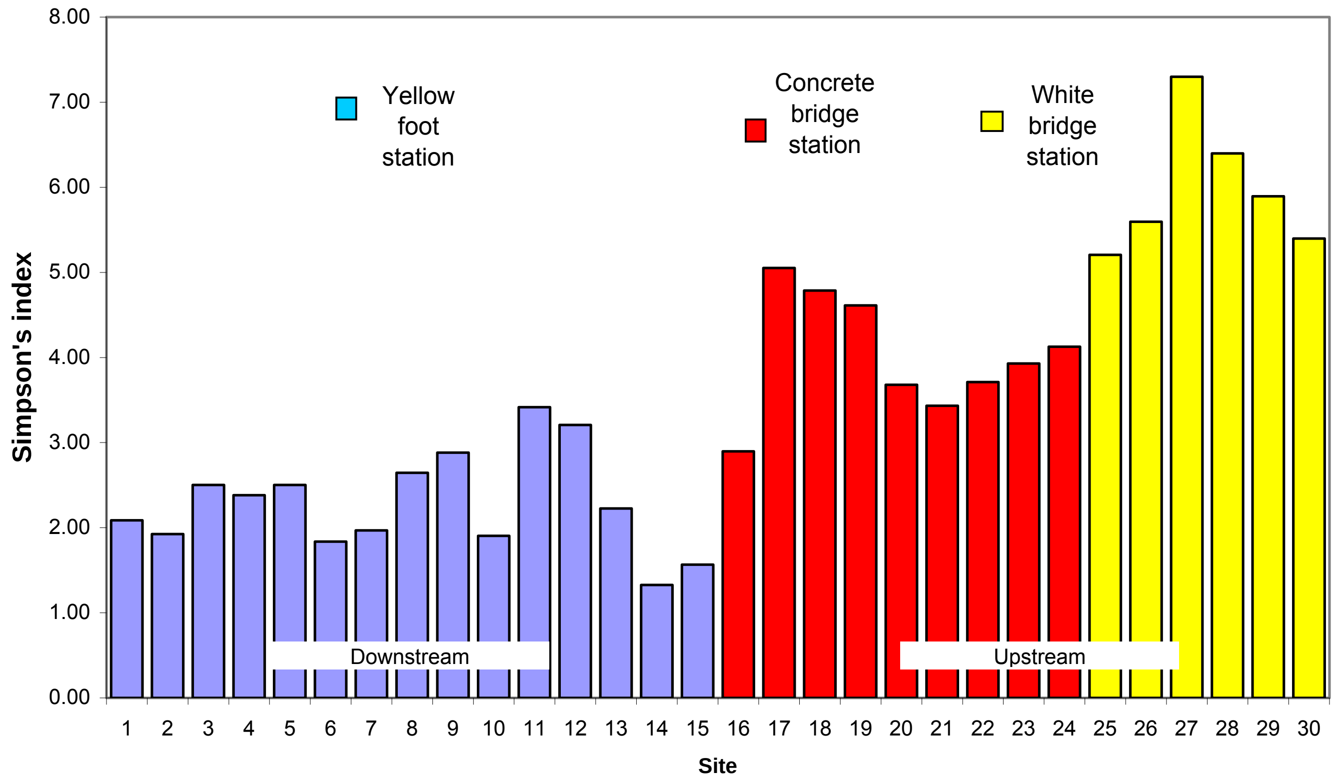

Figure 4.5.2. Simpson's index values of Diversity $\{D_s=\Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 6-8 October 2003.

4.5.3 Sampling period: 16-18 February 2004

There was an increase in both means of Yellow foot (4.18) and Concrete bridge station (4.34), but a drop in mean diversity values for White bridge station (5.71) compared with the previous sampling period (Fig.4.5.3). The most diverse site for the Yellow foot station was Site 14 (7.51), Site 19 (7.08) for Concrete bridge station and Site 29 (7.91) for White bridge station. The minimum values were at Site 1 (2.19) for Yellow foot station, Site 24 (2.43) for Concrete bridge station and Site 28 (4.22) for White bridge station.

4.5.4 Sampling period: 1-3 March 2004

There was an overall rise in diversity during this period, with Yellow foot station having a mean of 4.98, Concrete bridge station 10.06 and White bridge station 7.01 (Fig.4.5.4). The most diverse site for Yellow foot station was Site 7 (6.63), for Concrete bridge was Site 23 (17.5) and Site 28 (9.11) for White bridge station, while minimum values were at Site 1 (1.33) for Yellow foot station, Site 24 (4.6) for Concrete bridge station and at Site 26 (4.8) for White bridge station.

4.5.5 Sampling period: 15-17 March 2004

There was an overall drop in diversity from the previous sampling period, with a mean value for Yellow foot station of 4.25, Concrete bridge station 5.61 and the White bridge station of 5.06 (Fig. 4.5.5). The most diverse site for Yellow foot station was Site 10 (6.38), for Concrete bridge Site 21 (9.46) and Site 28 (6.62) for White bridge station, while minimum values were at Site 13 (2.82) for Yellow foot station, Site 18 (2.91) for Concrete bridge station and Site 30 (3.11) for White bridge station.

4.5.6 Sampling period: 29-31 March 2004

There was an overall increase in diversity from the previous sampling period (Fig. 4.5.6). Yellow foot station had a mean of 5.24, Concrete bridge station of 6.67 and White bridge station 5.66. The most diverse site for Yellow foot station was Site 15 (6.05), for the Concrete bridge Site 19 (8.77) and Site 28 (6.25) for White bridge station, while the minimum values were at Site 10 (3.3) for Yellow foot station, Site 24 (3.47) for Concrete bridge station and Site 27 (3.71) for White bridge station.

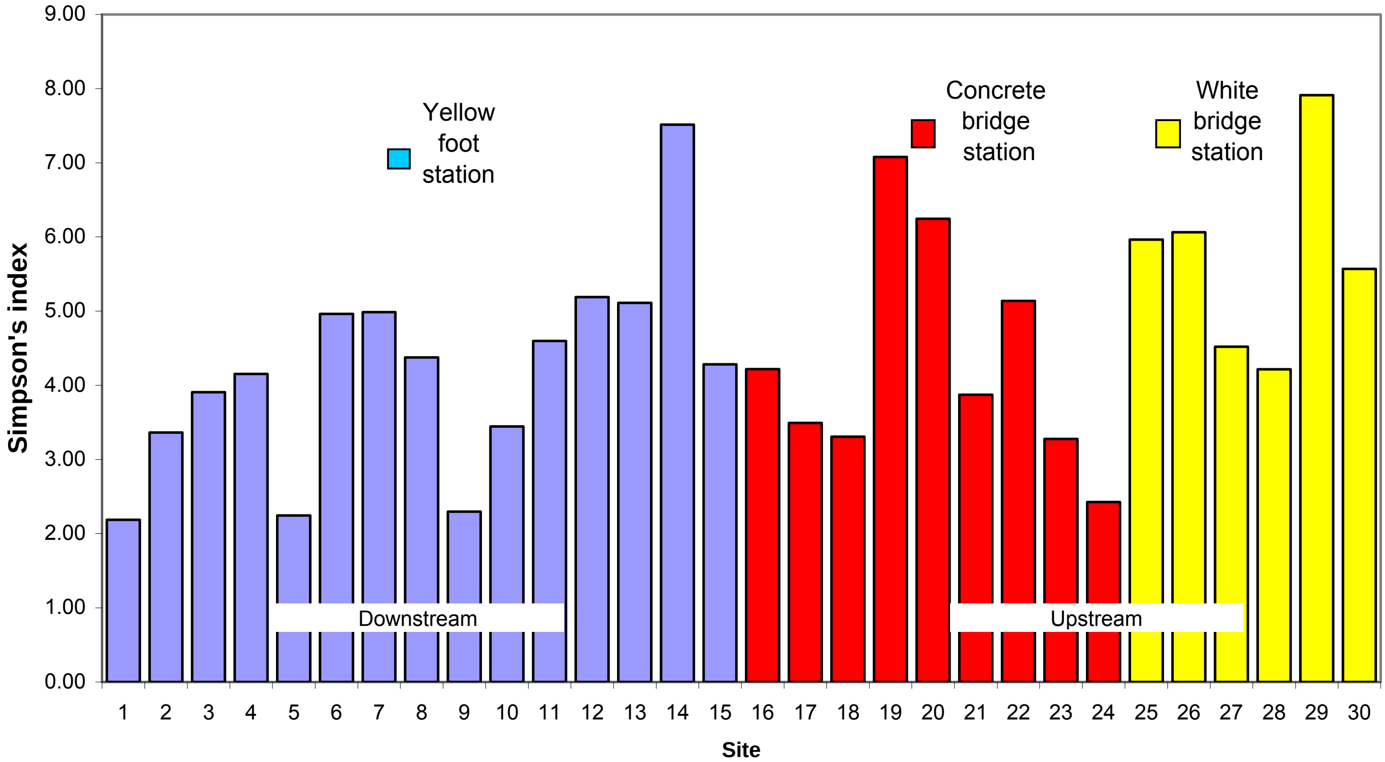

Figure 4.5.3. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 16-18 February 2004.

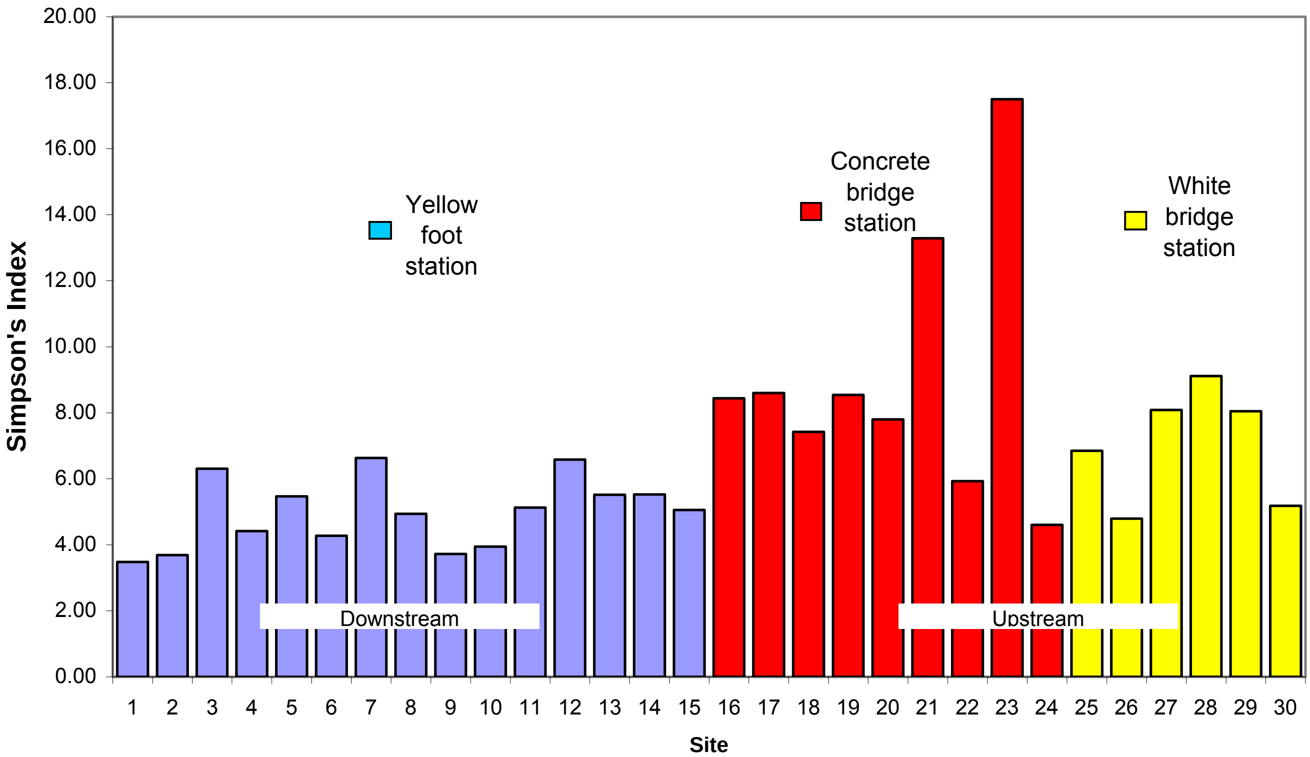

Figure 4.5.4. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 1-3 March 2004.

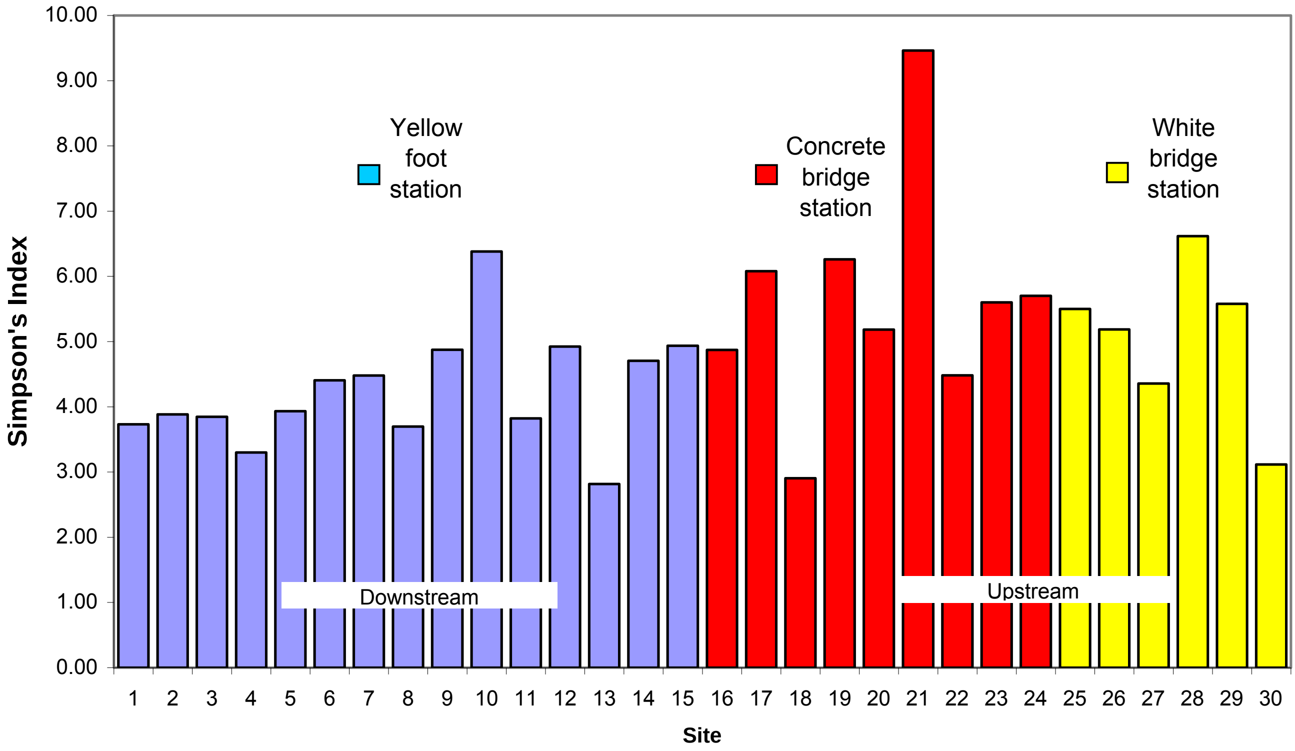

Figure 4.5.5. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 16-18 March 2004.

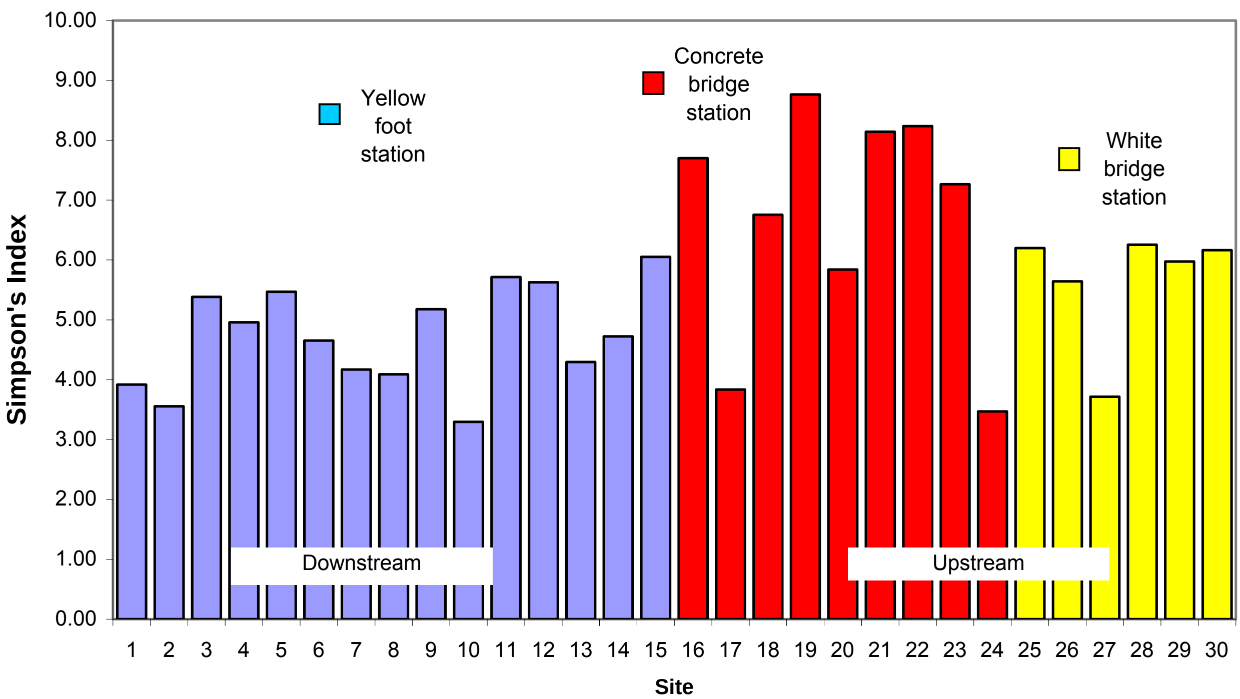

Figure 4.5.6. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 29-31 March 2004.

4.5.7 Sampling period: 10-12 May 2004

The mean for Yellow foot station was 3.62, for Concrete bridge station 5.61 and White bridge station 5.52 (Fig. 4.5.7). The most diverse site for Yellow foot station is Site 13 (5.37), for Concrete bridge Site 20 (6.99) and Site 30 (6.68) for White bridge station, while the minimum value was Site 9 (1.99) for Yellow foot station, Site 17 (2.93) for Concrete bridge station and Site 29 (4.21) for White bridge station.

4.5.8 Sampling period: 21-23 June 2004

There was an increase in diversity at Yellow foot and Concrete bridge stations from the previous sampling period, but a decrease for White bridge station (Fig. 4.5.8). The mean for Yellow foot station was 3.76, for Concrete bridge station 5.77 and White bridge station 5.00. The most diverse site for Yellow foot station was Site 5 (7.06), for Concrete bridge is Site 18 (8.43) and Site 29 (6.31) for White bridge station, while the minimum value was at Site 3 (1.92) for Yellow foot station, Site 16 (4.31) for Concrete bridge station and at Site 27 (3.96) for White bridge station.

4.6 SHADE

4.6.1 Temperature

The three shade divisions (Fig. 4.6.1)(>66%, 33–66% and <33% canopy cover) showed a peak in summer and a low in winter. During all sampling periods it was the <33% that showed the highest mean. The 33-66% division has the lowest mean for all sampling periods, except during October and June. At both these sampling periods there is very little difference between the three divisions. In all sampling periods, the >66% division canopy cover indicated it to be the most variable.

4.6.2 Conductivity

The three shade divisions (Fig. 4.6.2) (>66%, 33 – 66% and <33% canopy cover), showed a peak February to mid-March, with the lowest mean values in winter. Over the entire sampling periods the 33-66% division canopy cover showed the lowest means, while the >66% cover showed the highest values in the first two sampling periods, when it shifted to the <33% cover for the remaining sampling periods.

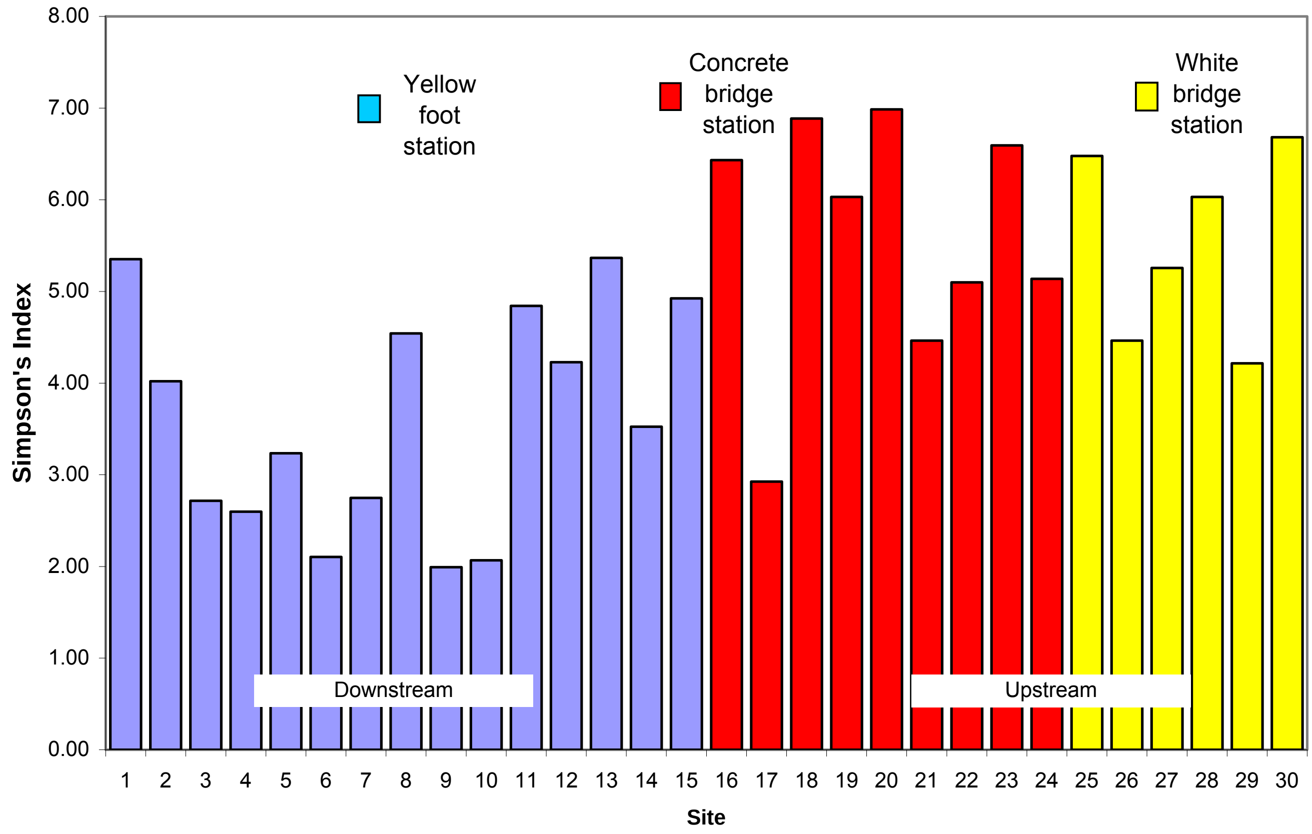

Figure 4.5.7. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 10-12 May 2004.

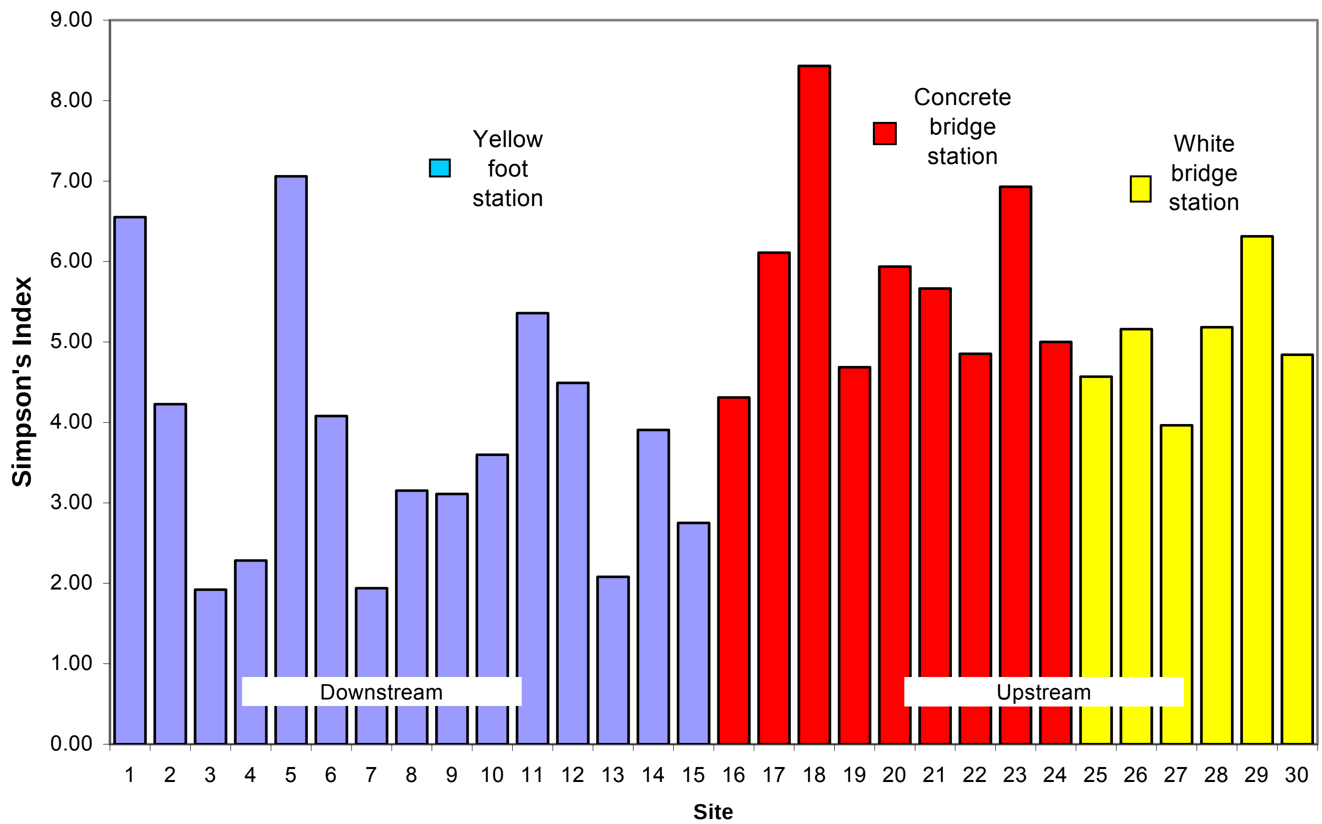

Figure 4.5.8. Simpson's index values of Diversity $\{D_s = \Sigma\ (n_i(n_i-1))/(N(N-1))\}$ for each of the sampling sites along the Eerste River, Jonkershoek, 21-23 June 2004.

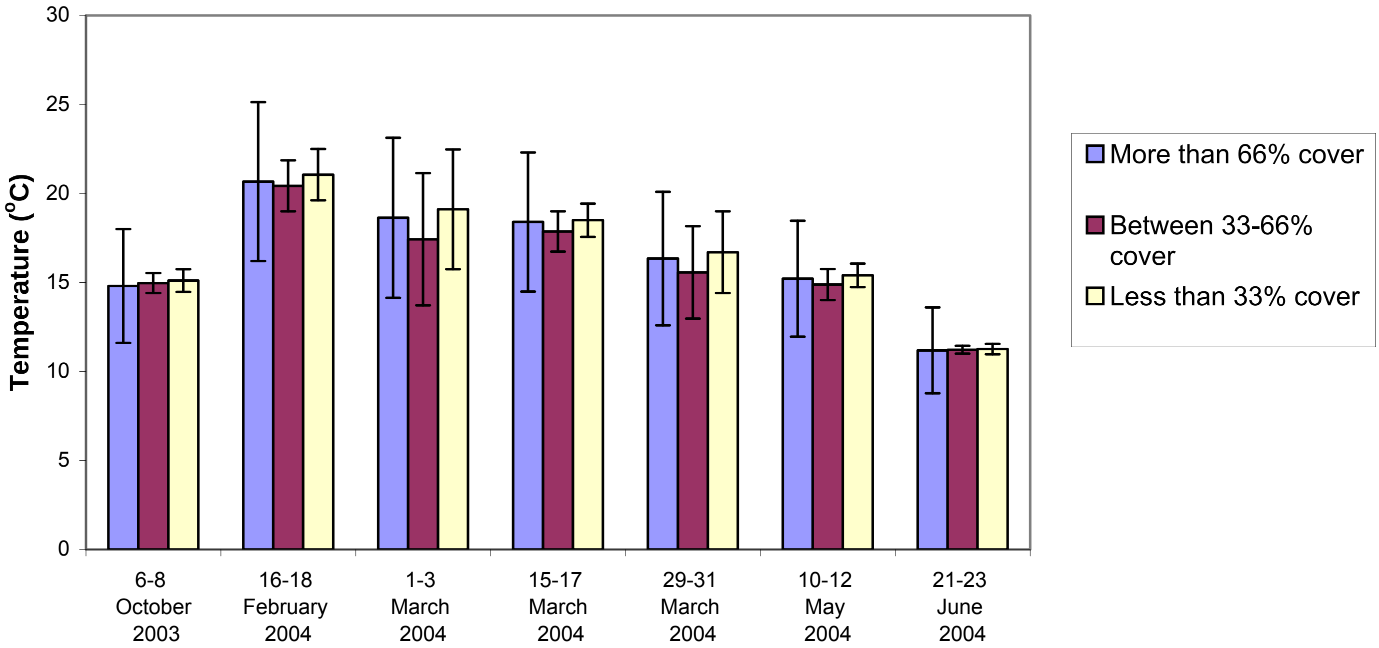

Figure 4.6.1. Mean Temperature in ^{o}C of sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek, for each of the sampling periods.

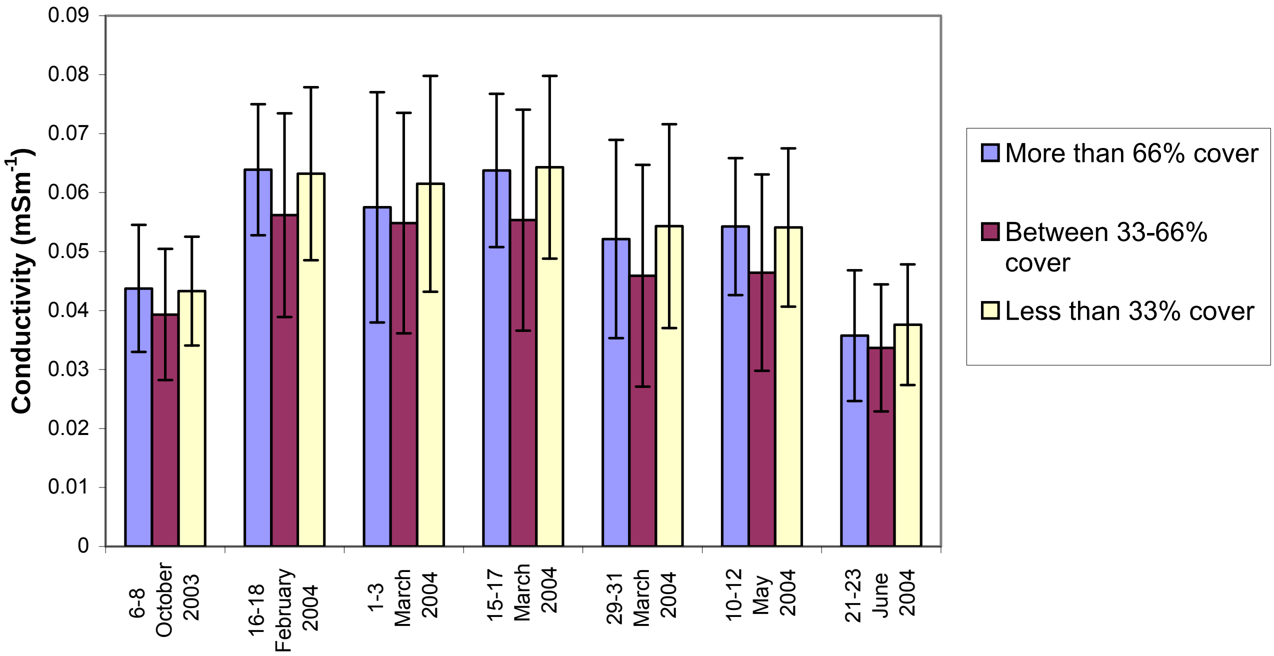

Figure 4.6.2. Mean Conductivity in mSm^{-1} between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek, for each of the sampling periods.

4.6.3 Dissolved Oxygen

Both dissolved oxygen in mg.ℓ$^{-1}$ (Fig. 4.6.3.1) and percentage dissolved oxygen (Fig. 4.6.3.2) are similar for all three shade divisions (>66%, 33 – 66% and <33% canopy cover). The maximum mean values were in October, with minimum mean values in March. Throughout the study period, the 33-66% division canopy cover showed the highest mean, with >66% showing the lowest mean, except for February, where the <33% cover division was the lowest.

4.6.4 Acidity level

The mean pH values (Fig. 4.6.4) for the three divisions (>66%, 33 – 66% and <33% canopy cover), were much more variable than the above mentioned abiotic variables. Early-March had the highest mean values for the >66% cover, while both other divisions peaked in May. The minimal means for both divisions with >33% cover were in winter, while the <33% division was in October.

In October, the 33-66% cover had the lowest mean. February, late-March and May showed a decrease in acidity with decrease of percentage canopy cover. In early-March, the >66% cover was the highest, with the 33-66% cover the lowest. Mid-March showed an increase in acidity with decrease of percentage cover, while June showed the <33% canopy to have the highest mean and the 33-66% cover the lowest.

4.6.5 SASS5 & ASPT scores

The South African Scoring System values (SASS5) (Fig. 4.6.5.1) indicated a peak for >66% cover in May, while both the other two divisions peaked in early-March. The minimal mean value was in mid-March for >66% and 33-66%, while the <33% cover minimal mean was in October. The 33-66% cover division was overall the highest, except in February where it was the <33% division that was the highest. The >66% division was the lowest in all the sample periods except for October and May.

ASPT (Fig. 4.6.5.2) gave the same pattern as the SASS5 scores, except for May, when the >66% cover was the lowest rather than the <33% division.

4.6.6 Number of individuals

The mean number of individuals (Fig. 4.6.6) in the > 66% division, showed a maximum in June, with minimum in early-March. The 33 – 66% division peaked during May with a

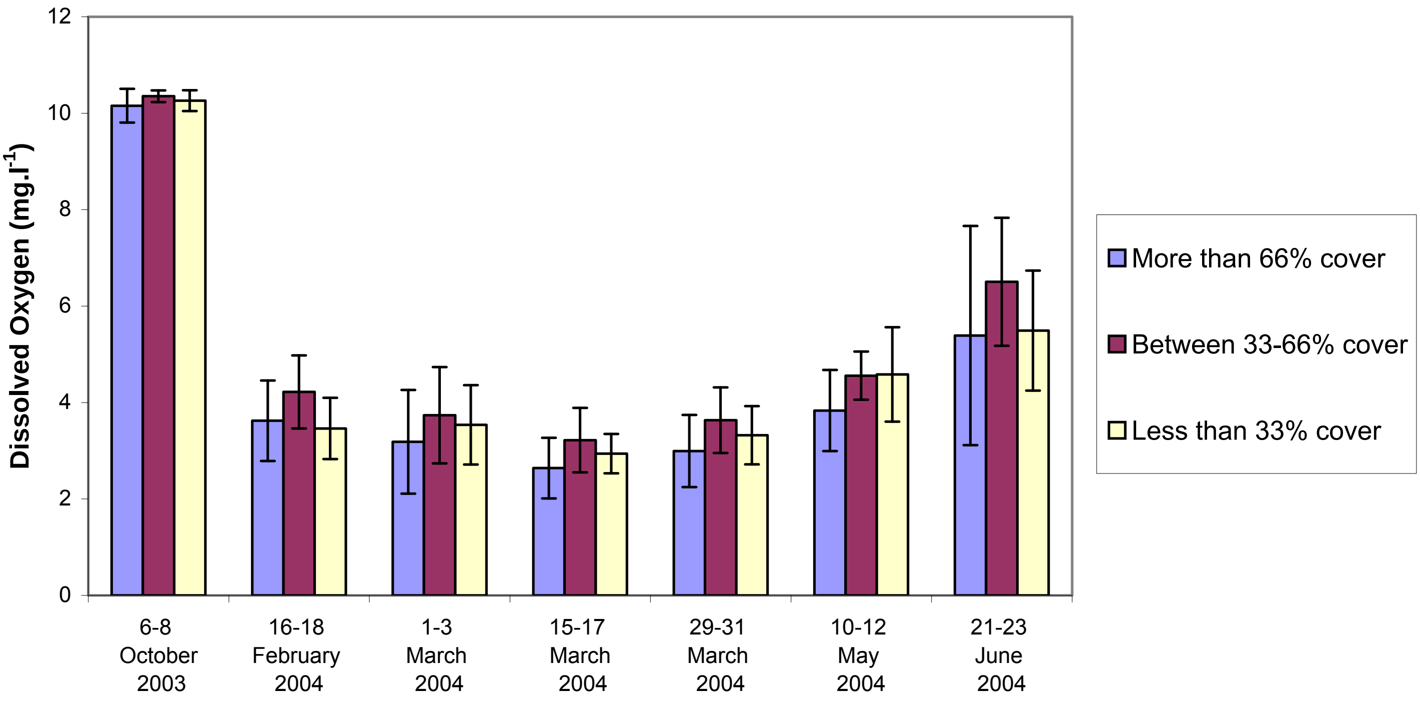

Figure 4.6.3.1. Mean Dissolved Oxygen in $mg.l^{-1}$ between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

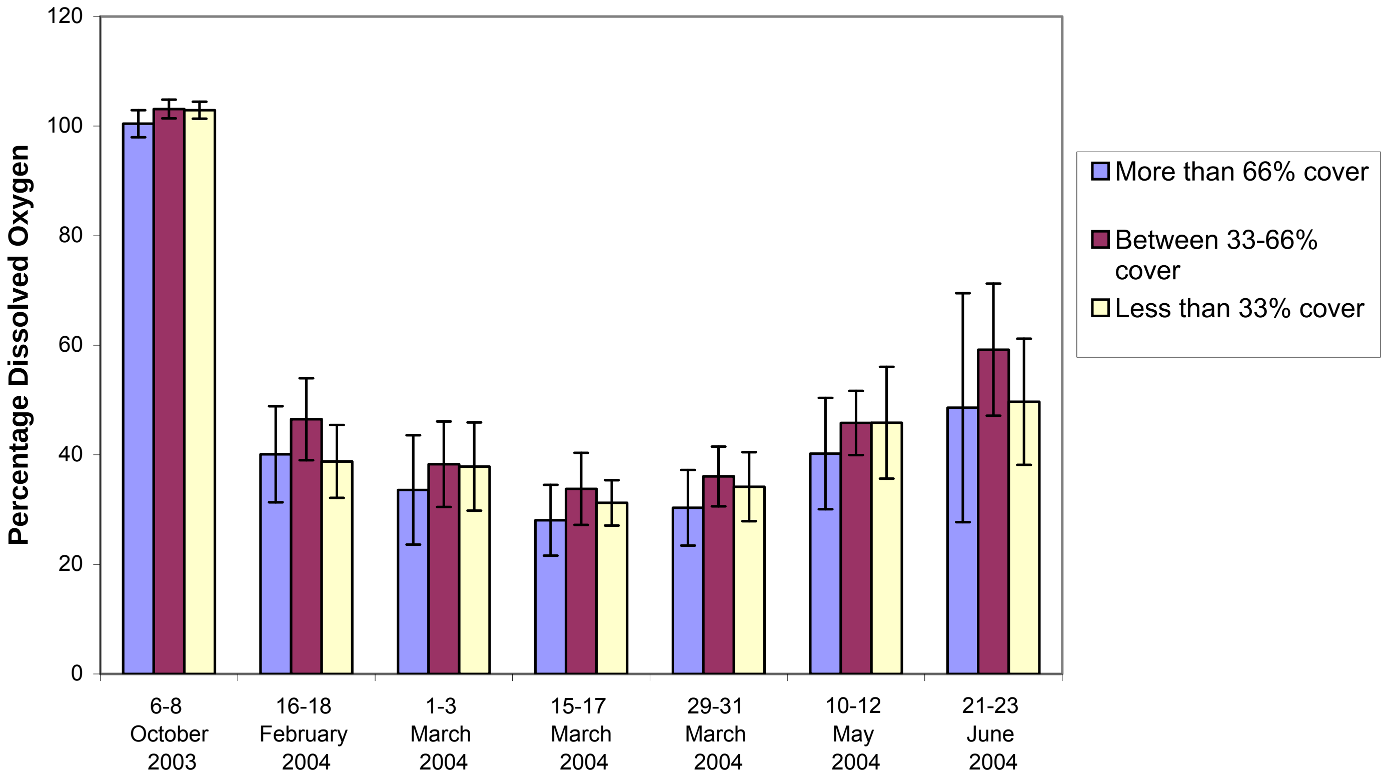

Figure 4.6.3.2. Mean Percentage Dissolved Oxygen between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

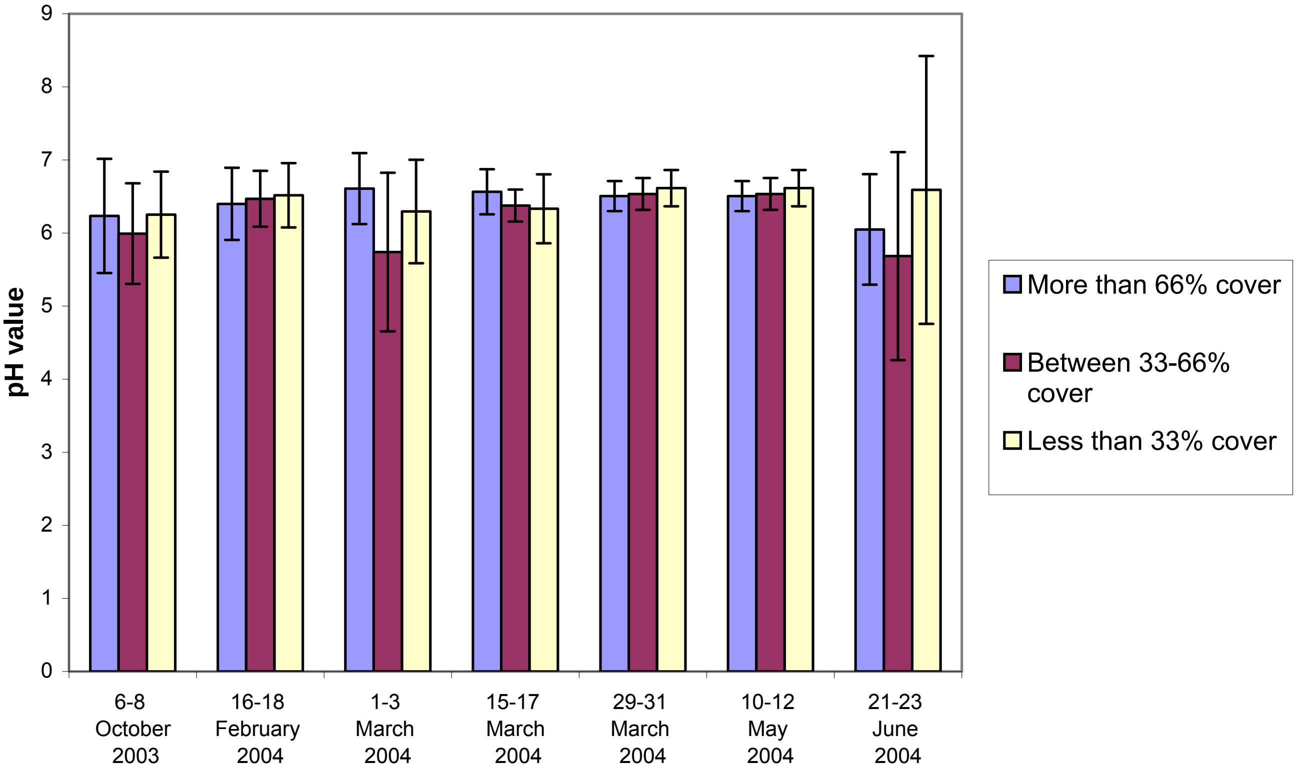

Figure 4.6.4. Mean pH value between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

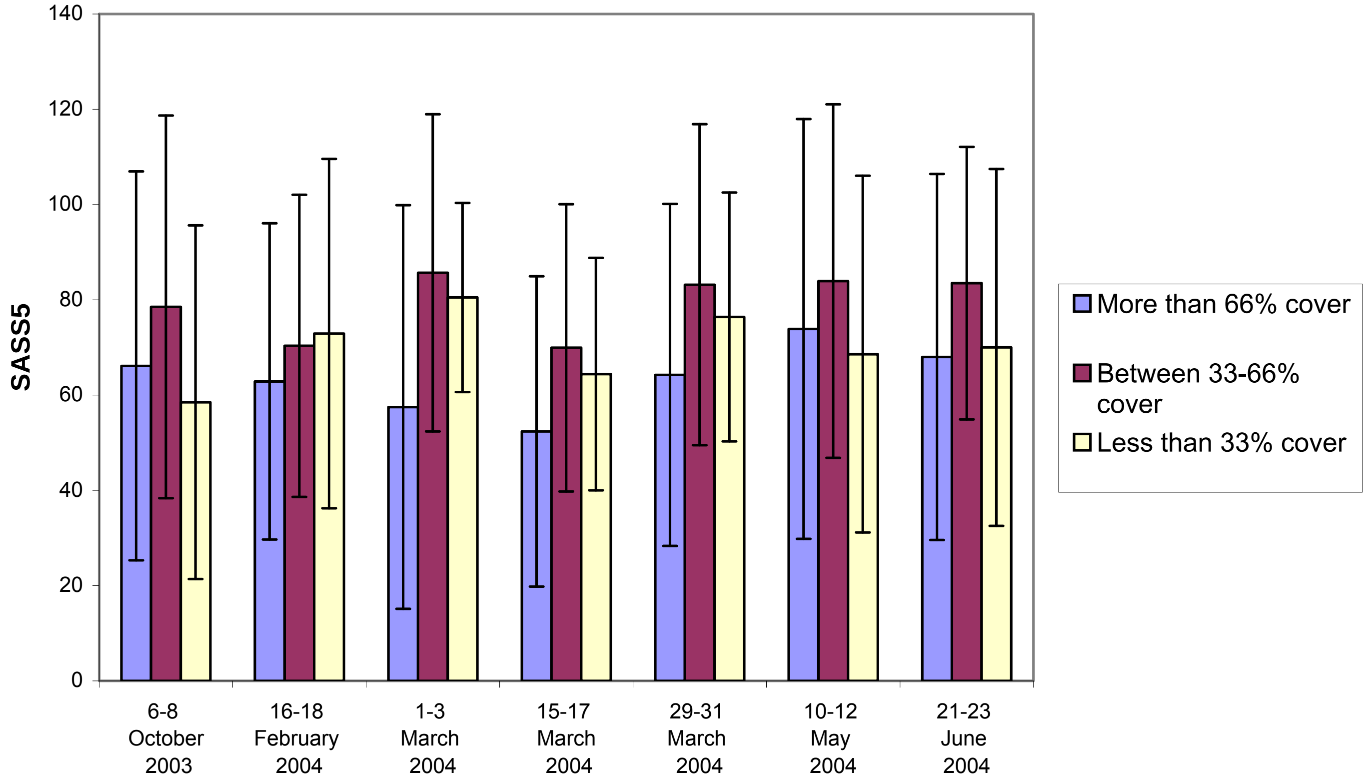

Figure 4.6.5.1. Mean SASS5 score between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

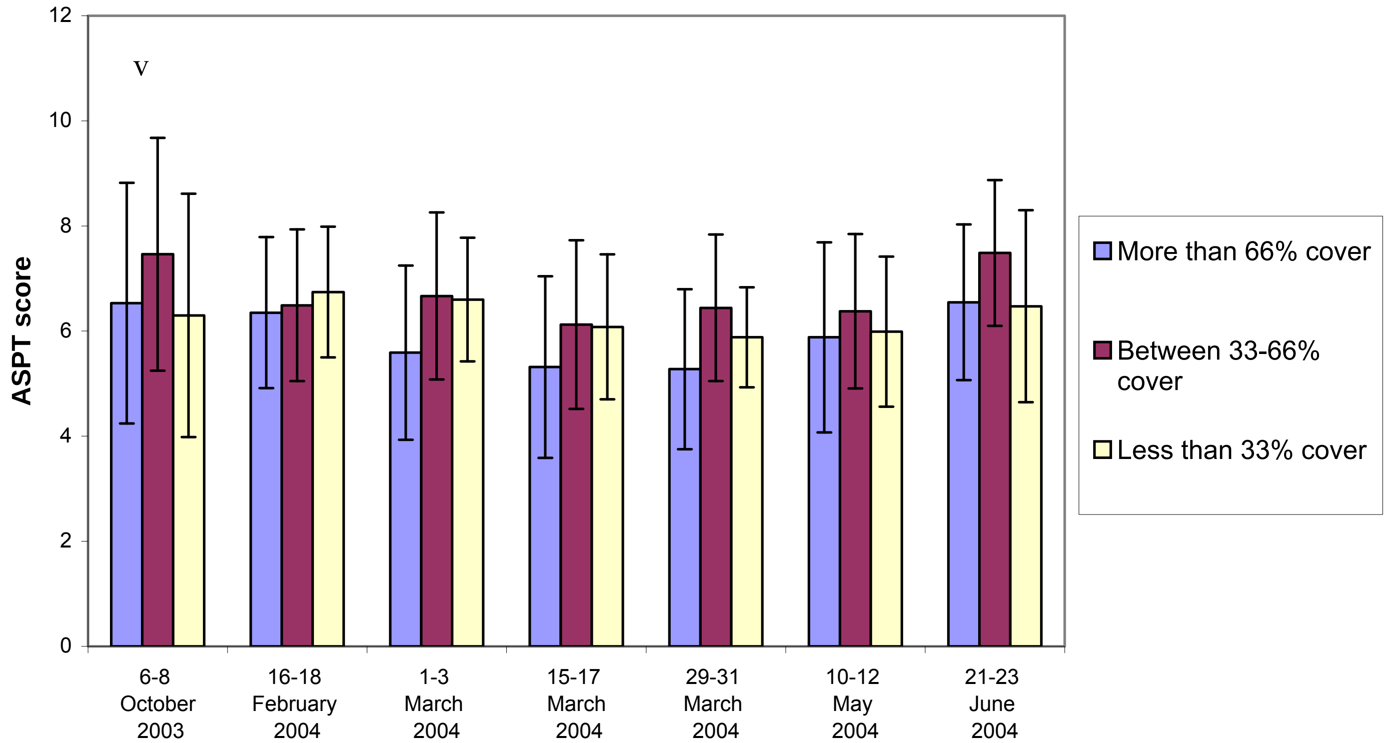

Figure 4.6.5.2. Mean ASPT score between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

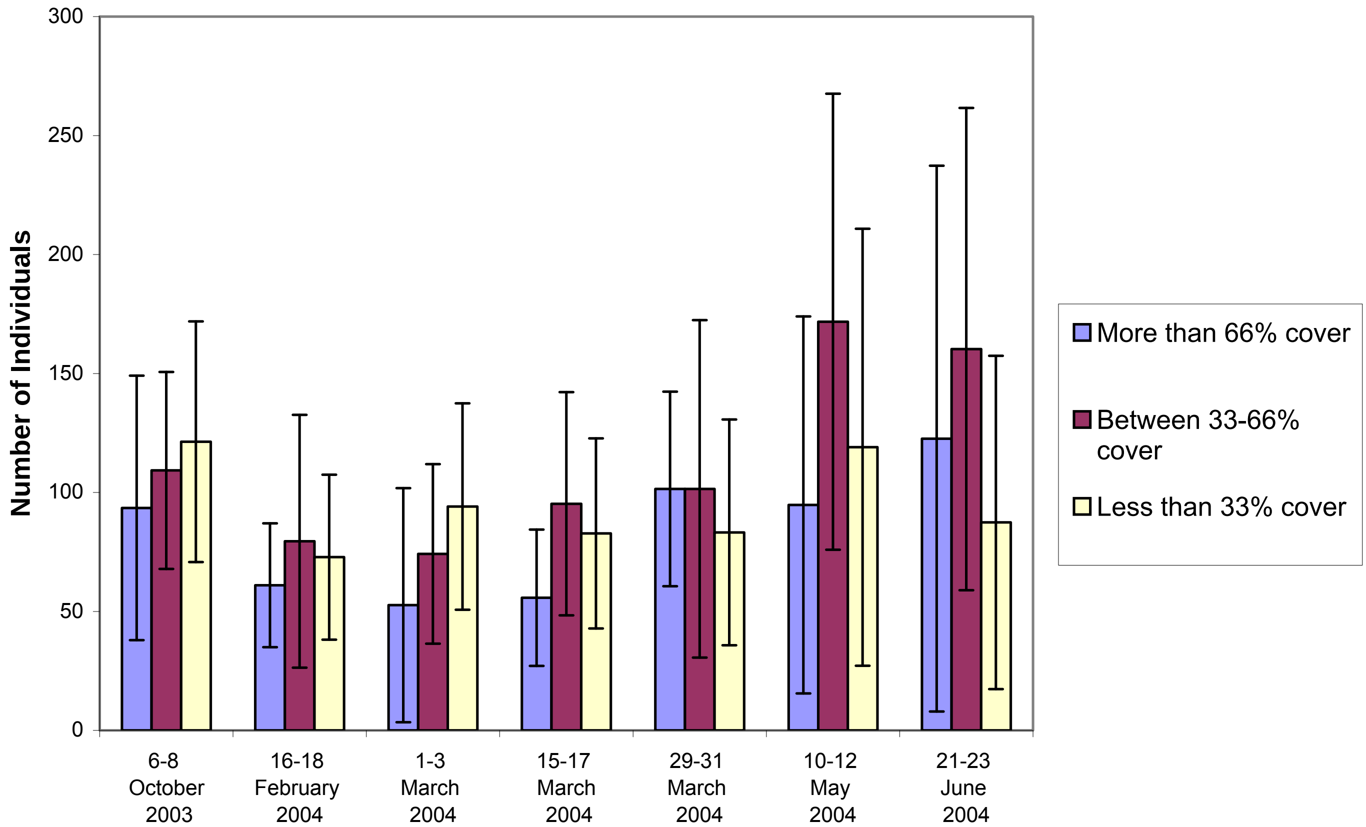

Figure 4.6.6. Mean Number of Individuals between sites with a cover less than 33%, between 33 - 66% and more than 66% along the Eerste River, Jonkershoek for each of the sampling periods.

minimum in early-March. For the <33% division, the maximum was in October, and minimum during February. For October and early-March there was an increase in abundance with a decrease of percentage canopy cover, while the rest showed greatest abundance in the 33-66% division. During all the sampling periods, the >66% cover division tended to have least abundance, except for late-March and June.

4.7 REGRESSION ANALYSIS

Neither of the regression analysis done for the Number of Taxa nor Number of Individuals indicated any significant correlation towards the Temperature, Conductivity, Dissolved Oxygen or pH values.

Chapter Five

DISCUSSION

5.1 BIOMONITORING OF WATER QUALITY

Rivers form the core of our freshwater systems and provide civilization with potable water. The human race has evolved along with this resource, building villages on its sides, using its floodplains to grow crops, its pathways to define the borders of territories, their flow for movement and transportation, as reservoirs for food, but most importantly for clean, fresh drinking water. With this progression, the human race has learned how to manipulate the river system to its own benefit by altering the flow, constructing barriers, impoundments, gabions, walls and using it for canalisation and waste disposal. Changes of abiotic and biotic factors along the longitudinal organization of a stream topography, have profound effects on, or have corresponding counterparts in, almost every aspect related to running waters, be they abiotic factors like stream flow, sediment composition, temperature regime, solute or particular freights, or be they biotic, including flora and fauna distribution or numbers (Vannote *et al.*, 1980).

Removal of streamside vegetation results in a number of changes, including higher water temperatures, altered channel structures due to reduced inputs of woody debris, fewer inputs of leaf litter and less retentiveness. Higher rates of soil erosion and runoff of fertilizers, pesticides and herbicides are common problems of agriculturally impacted rivers, while municipal and industrial wastes are important pollutants of rivers in urban areas (Allan, 1995). The replacement of indigenous vegetation by agricultural crops causes a shift from heterotrophy to autotrophy. Agriculture furthermore increases pressure on irrigation, which can lead to nutrient enrichment from return flows (Madikizela, 2001), due to fertilizers and animal wastes, and also by increasing soil erosion, which particularly affects the transport of phosphorus (Allan, 1995). Manipulation is the tool that made humans the most advanced species on earth, but with that power came responsibility.

Since water quality, or more directly, the pollution of water, is often transient and unpredictable, biological monitoring may be more appropriate than traditional chemical evaluation of water quality to assess contamination of aquatic ecosystems (Guèrold, 2000; Basset *et al*., 2004). The purpose of biological assessment is to characterize the status of water resources and to monitor trends in the condition of biological communities associated with anthropogenic perturbation (Resh *et al*., 1995). Any river site being assessed can then be compared with similar least-impacted ones, so as to provide a reference condition of how far removed from natural the site might be (King & Schael, 2001). In support of this approach, is the "River Continuum Concept" (Vannote *et al*., 1980), the "Concept of serial discontinuity" (Stanford & Ward, 1983) and the "Hypothesis of stream hydraulics as determinants for the distribution of organisms" (Statzner & Higler, 1986).

Water quality monitoring was traditionally based on the assessment of physical and chemical parameters, and is justified by the fact that some species can indicate the condition of water in which they live, making short and long-term monitoring more comprehensive and cost effective (Madikizela, 2001). A number of bioindicators and biotic indices and scores based on benthic macroinvertebrates, diatoms, fishes, aquatic and riparian vegetation in relation to physicochemical parameters have been applied in the assessing of water quality (Iliopoulou-Georgudaki *et al.,* 2003). Extensive literature on these groups and methods is available (Sladecek, 1973; Hawks, 1979; Persoone & De Pauw, 1979; Wright *et al*., 1984; Moss *et al*., 1987; Barbour & Stribling, 1994; Barbour *et al*., 1995; Resh *et al*., 1995; Iliopoulou-Georgudaki *et al*., 2003). Among the bioindicators used, the benthic macroinvertebrates seem to be the most reliable (Rosenberg, 1998; Iliopoulou-Georgudaki *et al*., 2003). Studies of the distribution of macroinvertebrates in lotic environments have also assisted in evaluations of environmental quality (Rosenberg & Resh, 1993; Roque *et al*., 2002).

Earlier physical and chemical monitors, consisting of mainly measures of temperature, dissolved oxygen, conductivity, nutrient levels and pH, were extensively used to detect pollution levels at a given location (Dickens & Graham, 2002). However, such monitoring instruments are very expensive, can be used at a limited number of sites and no distribution patterns or comparative data can be achieved

(Swaminathan, 2003). Therefore the data are only as good as the method and instruments used (Dickens & Graham, 2002).

Almost all rapid bioassessment approaches use techniques that evaluate assemblages of benthic macroinvertebrates at reduced costs relative to those associated with traditional, more rigorous, assessments (Resh & Jackson, 1993). The effort of benthic analysis is less with rapid assessment because of three specific features. First, a relatively large sample consisting of two to several collections is taken instead of several individual replicates. Secondly, a standardized sub-sampling procedure is used, which both reduces the number of organisms processed and provides a relative consistent unit of effort for the processing of all samples. Thirdly, the results of surveys can be summarized in ways that are understandable by non-specialists (Resh *et al*. 1995). The advantage of monitoring using of bioindicators is that biological communities reflect overall ecological quality and integrate the effects of different stressors, providing a broad measure of their impact and an ecological measurement of fluctuating environmental conditions, especially when assessing toxicant pollutants (Iliopoulou-Georgudaki *et al.* 2003).

In the past three decades, the study of aquatic insects has greatly advanced. Not only have research studies indicated the important role played by larvae of aquatic species in the breakdown of terrestrial leaf litter and the pathways by which the plant energy is incorporated into the tissue of fishes, birds and other vertebrates, but a host of further applied research has shown the importance of aquatic insects in the spread of diseases and in the reconstruction of past environments on earth. More recently, aquatic insects are increasingly being used to test many hypotheses in contemporary ecological theory, but most dominantly are being used in the biological assessment of water quality (Williams & Feldmate, 1992).

Many species of macroinvertebrates are diagnostic of certain kinds of habitats and specific water quality. They are known as indicator organisms, meaning organisms that become numerically dominant only under a specific set of environmental conditions (Mackie, 1998) and by their presence, abundance and activities, reveal something about the state of the ecosystem in which they are found. Whether in a state of deterioration or amelioration, they can suggest that ecosystemic processes are operating according to expectations within normal bounds. In the latter case, the species, taxa or guilds used may be indicators of ecosystem health (Kevan, 1999). The most common usage of benthic organisms is to indicate water quality, especially the trophic status of lakes, calcium hardness, alkalinity, pH and conductivity (Mackie, 1998).

Benthic macroinvertebrates are recognised as valuable organisms for bioassessments, largely due to their visibility to the naked eye, ease of identification, rapid life cycle often based on the season, their mainly sedentary habits (Dickens & Graham, 2002) and the potential database consisting of millions of species (Hammond, 1995).

Biomonitoring systems for the aquatic habitat not only make use of macroinvertebrates, but also vertebrates like osteichthyes and amphibians (Karr, 1981; Plafkin *et al.,* 1989; Oberdorff & Hughes 1992; Resh *et al*., 1995), micro-organisms like diatoms (Iliopoulou-Georgudaki *et al*., 2003), as well as vegetation (Iliopoulou-Georgudaki *et al*., 2003; Swaminathan, 2003).

5.1.1 Factors influencing diversity

The concept of biodiversity has grown with the perception of its loss due to increasing human impact and mismanagement of the environment (Willson, 1988). Whether on a local, regional or global scale, reduced biotic diversity is associated with increased environmental stress and reduced environmental heterogeneity (Erwin, 1996). The concept of biodiversity implies that any environment is rich in different organisms and can be regarded as a system in which species circulate and interact. Structure, scale and features of the landscape also enter into the definition of biodiversity (Paoletti, 1999). Diversity is mostly influenced by disturbance and the most stable region with the greatest varieties of different habitats will indicate a higher level of diversity, while disturbed areas with fluctuations in the physical features will have a low diversity. It is also expected that with the decline of elevation in a riverine system the diversity increases, because a greater variety of food is available and circumstances are not as extreme. The later part of the system's diversity is expected to decrease drastically due to lack of habitat, sunlight and food as well as the increase of salinity.

Several biotic factors, such as genetic diversity, can also affect biodiversity. Over the short term e.g. 1-10 years, biodiversity can fluctuate as different gene pools are randomly selected through short-term changes in the environment. However over the long term e.g. decades to centuries, biodiversity has declined due to both direct and indirect factors, such as habitat loss, habitat fragmentation, over-exploitation, pollution and the introduction of alien species (Mackie, 1998).

For most taxa in this study, there was an increase in diversity with the decline of elevation along the natural stream between the White bridge station and the Concrete bridge station. However, there was a drastic decline downstream. This is contrary to the expectation and suggests that the reservoir is in some way playing a major role in altering the aquatic fauna.

5.1.2 Clustering of sites

Roque *et al*. (2002) indicated that the study of the distribution of macroinvertebrates, considering multiple hierarchic scales and incorporating different spatial dimensions to assess the role of disturbance in aquatic environments, could contribute to conservation, environmental evaluation and improvement of analytical tools in ecology.

The Bray-Curtis analysis is expected to cluster sites with comparable habitats with similar physical and chemical components. It reacts on the base that a species is bonded to a specific niche that correlates with the habitat it occupies. Therefore the species presence will give groupings of similar circumstances.

The study here indicated differences between the three sampling areas for temperature, conductivity, pH, diversity and in some cases also dissolved oxygen. The region downstream from the reservoir showed the greatest differences, with the differences between the two regions upstream from the reservoir indicating no significant differences.

Shade also played a role in distribution, probably indirectly due to temperature and the amount of algal growth.

5.2 SOUTH AFRICAN SCORING SYSTEM

SASS is a biomonitoring system adapted for South African rivers to give an indication of water quality. This is done by looking at the macroinvertebrates present in the system and adding up a value derived from the species tolerance to pollution, with the most sensitive species having high scores while the most tolerant providing low scores.

The system ideally is sufficient for rapid assessments and clearly indicates disturbances in the system, but lacks factors seen in other bioassessment in a predictive phase as found in other studies (Rosenberg & Resh, 1993; Metcalf & Smith, 1994; Resh, 1995; Dickens & Graham, 2000). For example, factors to predict the expected effect of impacts before development occurs, followed by a monitoring and assessment phases, to measure and interpret environmental effects during construction and after the development has been completed (Rosenberg, 1981). Furthermore, it does not consider the rarity of the species in question, as well as the influences of factors other than pollution on the taxa abundance.

5.2.1 Prediction

Prediction would be a valid input in evaluation of the monitoring system especially in the scenario for South Africa as a whole. South Africa is a large country with eight climatic regions based on vegetation (Hodges, 1986). Each region has its own ecology that differs not only in the vegetation present, but also in climate and seasonal changes. These factors play a role in the stability of the riverine system as well as in the food sources available in the river system, thus indirectly influencing the invertebrates that can be present in the system. Therefore, it would have been profitable to have a system for each scenario rather than a broad scale system. Furthermore, predictions could be made according to the type of study area, giving a expected scenario where by adaptations could be made to the sampling method. Unfortunately, insufficient information is available on the ecology of macroinvertebrates to make such a study successful.

5.2.2 Rarity

The rarity of a taxon is deleted from the methods due to the difficulty of distinguishing between true and apparent rarity. In most biomonitoring systems, rare species are perceived as important in aquatic systems and are usually retained for the analyses, since their preservation is often the ultimate aim (Lenat & Resh, 2001; Basset *et al*. 2004).

5.2.3 Factors influencing the distribution of macroinvertebrates

Many ecological factors affect the distribution of aquatic macroinvertebrates. This includes landscape, physical and chemical features of water, the amount and type of aquatic vegetation, whether the water is stationary or flowing, weather and seasonal changes, as well as the life cycle and ecology of the species. Some species can tolerate a broad range of conditions, while others are very sensitive to their environmental conditions (Corbet, 1999).

5.2.3.1 Landscape

Considering a wide spatial scale, several features of the landscape can influence the distribution of the organisms. According to Richards *et al*. (1996), geologic attributes in combination with land use apply expressive influences in and on the structure of habitats in streams and, consequently, their biota (Roque *et al*., 2002).

In riverine systems, the variables are variations in stream order and the corresponding changes that occur downstream, as revealed in the River Continuum Concept (Vannote *et al*., 1980). These include substrate types, water velocity, depth and width of streams and sediment loads. All these physical attributes vary in relation to stream order, from coarse substrates (boulders, rocks, etc.) in clear, cold, well oxygenated water and narrow widths and shallow waters of lower order streams to fine sediments (e.g. gravel, sand, silt, etc.) in more turbid, warmer, less oxygenated water and wider and deeper waters of higher stream orders (Mackie, 1998).

5.2.3.2 Substrate types

A diversity of substrate types can be found at almost any stretch of river or stream, with each substrate in itself complex (Allan, 1995). This division is based on the size of abiotic substratum particles (King & Schael, 2001). However, even in this seemingly straightforward situation, organic detritus is found in conjunction with mineral material, and can strongly influence the organism's response to substrate (Resh *et al.*, 1995). In essence, the substrate includes everything on the bottom or sides of streams or projecting out into the stream, not excluding a variety of human artefacts and debris, on which organisms reside (Minshall, 1984). There are instances where the substrate is relatively uniform, as in sandy bottoms of low-gradient rivers, but usually it is very heterogeneous (Allan, 1995).

In general, diversity and abundance of benthic invertebrates increase with median particle size, and some evidence suggests that diversity declines with stones at or above the size of cobbles (Allan, 1995). But larger substratum also tends to have a larger abundance and variety of aquatic insects than a small type of substratum (Resh & Rosenberg, 1984; Minshall 1984; Resh *et al.*, 1995).

Fine sediment concentrations are usually higher downstream as rivers become wider and flow more slowly (Resh *et al.*, 1995). Sediment plays a major role in aquatic systems. It provides a habitat for many organisms, and most anthropogenic chemicals and wastes eventually accumulate there (Carpenter, 1988; Chapman, 1989; Chapman *et al.*, 1992; Crane *et al.*, 2000).

A stream with extensive silting will decrease the insect diversity as well as growth (Chutter, 1998) by altering biochemical conditions, food resources, respiratory diffusion gradients, and habitat space (Williams & Feltmate, 1992).

The amount of detritus trapped within the crevices is also likely to be important, and in this regard substrates of intermediate size are superior. A variable mix of substrates is expected to accommodate more taxa and individuals, and particle size variance usually increases with median particle size (Allan, 1995). Furthermore, diversity and abundance increase with substrate stability and the presence of organic detritus. Evidently the amount and type of detritus contained within the sediments is largely dependent on the size and mix of the mineral substrates (Allan, 1995). Other apparent factors, include the shape of the banks, the extent of floodplains (King & Schael, 2001), and surface texture, although it is difficult to generalize about their effects (Allan, 1995).

5.2.3.3 Water velocity

Water velocity and the associated physical forces collectively represent the most important environmental factor that indirectly affects aquatic insects. Biologists have long believed that water as a medium, and current as a force, strongly determine ecological distribution and shape, as well as anatomical and behavioural adaptations (Allan, 1995).

The speed of the current will determine the particle size, composition, and stability of the substratum (Resh & Rosenberg, 1984; Allan, 1995). Therefore, rapid water will have large sized substrata, which tends to be advantageous to insects, but it will limit the amount of captured organic material that will be negative to insect abundance (Resh & Rosenberg, 1984).

5.2.3.3.1 Currents

Current affects food resources via the delivery and removal of nutrients and food items. Current velocity also presents a direct physical force that organisms experience within the water column as well as at the substrate surface (Allan, 1995).

5.2.3.3.2 Direction of flow

Three fundamental types of flow characterize moving fluids: laminar, turbulent and transitional. In laminar flow, fluid particle movement is regular and smooth, and particles can be thought of as sliding in parallel layers with little mixing. Irregular movement with considerable mixing characterizes turbulent flow. Intermediate conditions are described as transitional. In fact, laminar flow conditions are so rare in aquatic environments that they are relevant primarily as a theoretical reference point (Davies & Day, 1998). As a consequence, quantification of flow conditions is problematic, and this is especially true for organisms dwelling on or near the substrate (Allan, 1995).

At the interface between a fluid and a solid the velocity of the two is identical, which means that water in contact with a non-eroding substrate has zero velocity (Allan, 1995). Because surface water can move quite rapidly, there must be a gradient in velocity as one approaches the bottom and sides of streams. This decrease in velocity with depth produces a region known as the boundary layer, which occurs where the speed of the current is no longer influenced by the presence of the stream bottom. In a shallow stream, the boundary layer may extend to the surface. Very close to the stream bottom there can be a laminate sub-layer where sheer stress is zero and flow is greatly reduced (Allan, 1995; Davies & Day, 1998). The complexities of flow around obstructions on and near the streambed are of particular importance, because most organisms of running waters live under these complicated near-bed conditions, and mostly in the middle of the water column (Allan, 1995).

5.2.3.3.3 Adaptations

Aquatic invertebrates possess anatomical features that enhance their ability to move about, or minimize their likelihood of being swept away. In some instances the functional benefits of such features are clear, while others are less obvious (Hynes, 1970). Direct attachment devices include silk and other sticky secretions, hooks and suckers seem appropriately identified as adaptations that aid them in maintaining position against the current (Picker *et al*., 2002, Davies & Day, 1998). Blepharoceridae larvae occur on smooth rocks in fast water, where their ventral suckers enables them to move against very high current velocities (Gerber & Gabriel, 2002). Simuliidae larvae are able to occupy high-velocity habitats by spinning a mat of silk onto a stone surface, to which they adhere with specialized prolegs. Circlets of outwardly directed hooks on both anterior and posterior prolegs aid the larva in this attachment. Should it be dislodged, an additional line of silk retains the larva to the substrate, allowing the animal to climb down the thread and re-attach itself (Williams & Feltmate, 1992). Silk is also used for attachment in other Diptera, and by Trichoptera to attach their pupal cases to stones while moulting (Allan, 1995). The body shape of animals exhibits a number of morphological adaptations that are viewed as adaptations to move about in current or avoid being swept away. A low vertical profile and streamlined shape are frequently observed in the running water biota (Williams & Feltmate, 1992; Allan, 1995). Dorsoventrally flattened or streamlined body shapes, such as of the Ephemeroptera and Plecoptera, might be adaptations to avoid or resist the pressures in flow (Allan, 1995).

5.2.3.4 Physical and Chemical features

5.2.3.4.1 Temperature

The temperature of running water usually varies on a seasonal and daily scale, and among locations due to climate, elevation, extent of streamside vegetation and the relative importance of groundwater inputs (Allan, 1995). The temperature of groundwater usually is within 1°C of the mean annual air temperature, and wherever groundwater inputs are important seasonal temperature variation will be slight (Allan, 1995). In nature, these temperature fluctuations during winter and summer keep the insect's ecology intact (Resh & Rosenberg, 1984).

In this study it was evident that the reduction in flow caused by the impounding of the water, allowed radiation to heat up the slower-flowing water downstream from the Klein Plaas dam. This area also naturally has less canopy cover due to the dieback of the vegetation from lack of the normal water supply, causing increased sunlight penetration. The increase of sunlight also caused an increase in microbial and algal activity that might also have had a further influence in the increase of the mean water temperature of the region below the dam. The highest temperatures were measured during the summer months when the day temperatures were highest, as well as day lengths being the longest. Even the activities and hydrology of the impoundment itself may have caused an increase in temperature in the dam, thus releasing warmer water into the system.

All aquatic macroinvertebrates are ectothermic and therefore temperature plays a significant role in their ecology (Resh & Rosenberg, 1984), and affecting their life processes including growth rates, life cycles and the productivity of the entire system (Allan, 1995).

Every aquatic species is restricted in its distribution to a certain range of latitude and elevation, and thus to a certain temperature range as well. Species found in cool climates typically are cool adapted and those of warm climates warm adapted. This does not necessarily mean that temperature preferences are the ultimate determinant of where a species will lives. It is equally likely that other factors determine range limits and the species has adapts to the conditions it experiences there. Nonetheless, temperature unquestionably sets limits on where a species presently can live (Allan, 1995).

Sweeney (1993) suggested that temperature affected growth and controlled the endocrine system of insects, which in turn determined the ultimate size and reproductive capacity of the adults. A low temperature decreased the metabolic rate of the insect, which in turn slowed its growth rate and reproductive ability (Resh & Rosenberg, 1984). The longer it took for an insect to reproduce, the fewer there were in a stream. Stunted growth also limited the number of progeny of the insect. High temperatures were expected to be advantageous to aquatic insects as it raised their metabolism, but this was not the case. Increased growth rate was a negative affect since it favoured the male insects (Resh & Rosenberg, 1984). The advanced emergence of the males limited their chances of reproduction. A study on ten aquatic insect species showed that thermopollution resulted in insect emergence as much as four to five months earlier than usual (Nebeker, 1971).

Variation in temperature was extremely important and if it did not occur as expected, it could negatively affect population and diversity of aquatic insects (Resh & Rosenberg, 1984). Studies have shown that normal fluctuations of temperature allowed eggs to hatch faster than at the mean temperature without fluctuations. This was especially important for eggs that diapauses before hatching. Most insects go into dormancy and out of it during the winter as a result of changes in water temperature (Ward, 1992). Many species of Chironomidae (Diptera) and Trichoptera can survived being completely frozen for over five months. In fact, many aquatic insects grew at or near 0^{o} Celsius (Ward, 1992).

Aquatic organisms may be particularly useful as biological indicators of modified temperature regimes or for modelling and monitoring temperature changes (Sweeney, 1993; Lowe & Hauer, 1999).

5.2.3.4.2 Dissolved oxygen

Aquatic insects require dissolved oxygen to breathe, so low concentrations of it will be fatal to them. Most dissolved oxygen accumulated in stream water by diffusion, which was a slow process unless it was accelerated by high water current ripples

(Ward, 1992). This study indicated that the dissolved oxygen values were the highest in areas with high flow during summer and autumn months when water levels were low, as upstream from the Klein Plaas dam. In these cases, there was a greater ratio of rapids and riffles, and the water surface was frequently disturbed by protruding rocks. These disturbances increased the mixing of oxygen into the water and thus increased the levels of dissolved oxygen in the system (Ward, 1992). Algal growth and plankton also played a role in oxygen levels, but more in slow flowing than in fast flowing water. This restoring method was much slower and less efficient than by surface disturbances.

In comparison, the low oxygen concentration at certain times of the year may be attributed to sediment loads, or photosynthesis rates and increasing temperatures (Palmer *et al*., 1990; O'Keeffe *et al.,* 1996).

5.2.3.4.3 Level of acidity

The underlying geology of the Eerste River catchment partly determines the natural water quality conditions expected in the system. In the upper catchment of the Eerste River, sandstone of the Table Mountain Series dominates. Waters draining such formations are typically pure, with low concentrations of dissolved substances (Dallas & Day, 1993). The naturally low buffering capacity of the water results in the pH being determined largely by vegetation type (fynbos), the decay products of which produce weak organic acids (Brown & Dallas, 1995).

The amount of leaf litter, especially from fynbos vegetation, influences the level of acidity, the more litter the higher the pH value. Another consideration is the quantity of water diluting a system. Furthermore, standing water also tends to go acidic over time if there is no inflow of freshwater. The experimental cage-culture trout farm situated in the Klein Plaas dam might cause a rise in the pH values of the river, by dilution due to the increase of water, but also by entering nutrition based food that are mainly alkaline.

Usually the acidity of a stream declines from the headwaters to the lower reaches (Ward, 1992). Most unpolluted streams have a pH value of 6.0 - 9.0 (Ward, 1992). The acidity level of pH below 6.0 is considered acidification of the stream.

Acidification usually results in low diversity and low productivity of aquatic insects (Resh & Rosenberg, 1984). Ward (1992), suggested that Ephemeroptera and Plecoptera species are incapable of surviving acidic levels lower than pH values of 6.0, but of this present study this was found not to be the case. In the present study the mean pH values in most cases were below these levels, especially upstream from the reservoir, where both Ephemeroptera and Plecoptera numbers were the greatest, but there appeared to be higher diversity upstream of the dam.

5.2.3.4.4 Organic levels

Organic detritus was found in combination with mineral material, and can strongly influence the organism's response to substrate (Resh *et al*. 1995). The amount of organic material directly affected the number of herbivorous insects and those again indirectly affected carnivorous insects. This was because most herbivorous insects fed on organic material. A low quantity of organic material created competition that will lowered herbivorous insect populations (Resh & Rosenberg, 1984).

Most salts and some nutrients, such as phosphates, were easily absorbed onto sediment particles, and could be carried with the flow downstream (Klotze, 1994). While model simulations demonstrated the association of water column dissolved oxygen, primary production, allochthonous organic matter, and the structure of the macroinvertebrate community, the quality and quantity of allochthonous carbon was shown to have considerable importance, not only as a food source but also as an oxygen sink (Spieles & Mitsch, 2003).

Normally, rivers did not become nutrient enriched, as they self purified during periodic floods and peak flows (Madikizela, 2001). In this system, most of the water was impounded during the rainy season and thus experienced nutrient enrichment in the areas downstream from the reservoir. Nutrients normally occurred in low concentrations in lotic systems (Chapman *et al.,* 1992), except when there was an external input (Madikizela, 2001), as was done by the experimental trout farm in the Klein Plaas dam or by the distribution of inorganic (e.g. phosphates) and organic (e.g. manure) nutrients on agricultural areas, some of which formed part of the runoff during heavy precipitation events (Mackie, 1998).

5.2.3.4.5 Vegetation

The riparian vegetation played an important role in the dynamics of the aquatic body and directly or indirectly influenced the processes and the communities of the system (Vannote *et al.*, 1980). Many macroinvertebrate species that were restricted to small streams reflected the ecological characteristics of the surrounding terrestrial community. In such streams, conditions for the larvae were affected by shade in summer or winter, the amount and periodicity of leaf-fall, and the distribution of local precipitation, all three of which were integrated with the type of climax community occupying the general area (Williams & Feltmate, 1992).

Invasive alien trees that were (after habitat loss) widely recognised as the second-largest threat to biodiversity (Walker & Steffen, 1999; Wilcove *et al*., 1998) could dominate riparian systems and changed the hydrology of the river (Le Maitre *et al*., 1996; Gőrgens & van Wilgen 2004). Samways & Taylor (2004) indicated that invasive alien trees form the major threat to Odonata species, by causing shrinkage of sunlit habitats for the adults.

5.2.3.4.6 Seasonal and weather changes

Seasonal variations have been reported in water quality parameters, such as conductivity, total suspended solids (TSS), pH, temperature and oxygen levels (Nelson & Roline, 1996). Furthermore, they could cause dieback of macrophytes, high and low water flows, scouring of backwaters and lush growth of problematic drifting vegetation (Hawking & New, 2002).

5.2.3.4.7 Life cycle and ecology

Species will adopted one of three life history strategies in order to live in stable and unstable environments: (i) r-selection; (ii) K-selection; or (iii) Bet-hedging strategy (Stiling, 1999). Stable environments were those in climates that were relatively constant or predictable, as in tropical climates. Unstable or unpredictable environments were characteristic of variable climates, such as in the temperate zones. Stable environments were characterized by species with a K-strategy, while fluctuating environments were characterized by species with a r-strategy. Advocates of r- and K- selection deal with models in which fecundity and mortality schedules fluctuate. The Bet-hedging strategy was advocated when fluctuations in these life history traits occur (Mackie, 1998).

5.2.3.5 Effect of pollution on the system

Pollution is a vague term used to describe changes in the physical, chemical or biological characteristics of water, air or soil, that can affect the health, survival, or activities of living entities (Williams & Feltmate, 1992). Organisms respond to pollution usually in one of two ways, acutely or chronically. Acute effects result in serious injury to, or death of, the organism shortly after exposure to high concentrations of a pollutant. Chronic effects are realized after exposure to low concentrations of a pollutant, the results of which appear over time.

The chemical nature and concentration of pollutants will usually change as a result of four natural processes: dilution, biodegradation, biological amplification and sedimentation. The rates at which these processes occured (particularly dilution and the oxygen-consuming process of biodegradation) varied directly as a function of the turn-over time of water in a system (Williams & Feltmate, 1992).

When released into the aquatic environment, many anthropogenic chemicals bind or adsorb onto particulate matter. Depending on river morphology and hydrological conditions, suspended particles with associated contaminants can settle along the watercourse and become part of the bottom sediments, often for many kilometres downstream of chemical sources. Many studies have demonstrated the remobilisation of contaminants from bed sediments (Hand & Williams, 1987; Cornelissen *et al*., 1997; Jeremiason, 1998; Zeng *et al*., 1999), which could thus elicit acute toxicity but, more frequently, chronic and sublethal responses in aquatic organisms (Landrum & Robbins, 1990; Burton *et al.,* 1996).

The dominant effect of pollution on the macroinvertebrate was the decline in numbers, but in the same cases the opposite is also possible. Water bodies suffering from industrial pollution were generally characterized by high densities of Chironomidae (Diptera), with an absence of Ephemeroptera and Plecoptera (Vigano *et al*., 2002). Furthermore, direct inputs of manure by cattle resulted in heavy growths of blue-green surface algae, attached green algae and submerged macrophytes

(Mackie, 1998), causing an increase in numbers of species feeding on above-mentioned vegetation, as soon as the concentrations of the pollution decline.

5.2.4 Invertebrate presence

In this study Ephemeroptera dominated in the Mountain stream zone and Diptera in the disturbed area downstream from the Klein Plaas dam. This could be ascribed to the fact that Ephemeroptera form a higher proportion of the total density in riffles instead off pools and the reverse can be expected for Diptera (Logan & Brooker, 1983). Both orders consist of species that are highly adapted to the aquatic system (Williams & Feltmate, 1992).

The seasonal changes and life cycles of the different species all played a part in the actual numbers found in the system. What was notable was that in species with a dominantly univoltine life cycle, a clear decline in numbers during the time the adult emerged from the water could be seen. This drop in numbers, especially in the Ephemeroptera and Diptera, triggered an increase in the numbers of the other taxa. This was probably due to the decline of competitiveness toward shelter and food resources.

Ephemeroptera numbers consisted mainly of the families Baetidae that preferred the upstream area but was also found in lesser numbers downstream from the reservoir in comparison to the Leptophlebiidae, which were basically only found in the area upstream from the reservoir. In contrast, Caenidae was mainly found in the downstream area at sites with an abundance of silt and sedimentation.

Diptera numbers were mainly supported by the Chironomidae for the downstream area and Simuliidae and Blepharoceridae, which were only found in high flowing rivers, and therefore upstream from the dam.

The above-mentioned phenomenon was expected with Baetidae and Simuliidae, both preferring habitats with riffles instead of standing water (Logan & Brooker, 1983). Furthermore, it was also known that heavily polluted habitats appeared to be dominated by Chironomidae, moderately polluted habitats by Chironomidae and Trichoptera, while unpolluted habitats contained Trichoptera, Plecoptera and

Ephemeroptera (Rosenberg *et al*., 1986). This suggested that in the present study the upstream area was in the unpolluted division, while the downstream area was within minimally polluted and disturbed categories. The proportion of Chironomidae in samples can also be a useful index of heavy-metal pollution (Winner *et al*., 1980).

Trichoptera numbers were divided between the cased Trichoptera of the families, Barbarochthonidae, Sericostomatidae, Glossosomatidae and Petrothrincidae, occurring in the high flowing waters of the sampling area upstream from the reservoir and the case-less Trichoptera of the families Ecnomidae and Hydropsychidae that occured throughout the system, but with significantly higher numbers downstream from the reservoir.

The larvae of Coleoptera families Elmidae, Gyrinidae and Helodidae were specialised for the harsher mountain stream zone (Gerber & Gabriel, 2002), and therefore occurred in high abundance upstream from the reservoir, while adult numbers were distinctively lower, with the families Hydraenidae and Hydrophilidae most abundant downstream.

It appeared that Notonemouridae (Plecoptera) was extremely sensitive to environmental changes, as indicated by their high abundance upstream from the reservoir, and virtual absence downstream from the reservoir.

Most Hemiptera were either lentic or slow-water lotic forms. They were all air breathers, and as such are more tolerant of environmental extremes than most other insects (Gerber & Gabriel, 2002). The water boatman, *Hesperocorixa*, and the water strider, *Gerris*, were among the few insects that could tolerate pH values less than 4.5 and were among the last to disappear when lakes and streams acidified (Williams & Feltmate 1992; Mackie, 2001). It was therefore understandable that the Hemiptera numbers were evenly spread from the more acidic areas upstream from the dam to the slower flowing areas downstream from the dam.

The distribution can be better associated with the feeding guilds that each family belongs to. Therefore shredders (Ephemeroptera, Plecoptera, Coleoptera and Crustacea) would be in abundant in areas with a high leaf litter presence, and indirectly a high canopy cover, as upstream. Grazers would be located mainly where algal growth is the highest, and therefore slow moving water with very little canopy cover. Filter feeders (Diptera and Trichoptera) need organic material to be broken down by mainly the shredders and thus would be more abundant further downstream, while predator numbers (Odonata, Hemiptera and Megaloptera) are dependent on the areas with the highest numbers of individuals, making them indirectly dependent on the environmental conditions of their prey.

The seasonal changes in the food sources can also influence invertebrate numbers. For example, during the autumn months leaf litter is most plentyful and therefore the shredders adapt their lifecycles toward, an abundance of larvae present in these time frames.

5.2.5 Average Score per Taxon

Of the various indices available the Average Score per Taxon (ASPT) is the most consistent over all biotopes. The guidance principle is to be aware of cases where habitat diversity is poor, that there will be less biotic diversity and consequently a lower SASS score. ASPT will be less affected (Chutter, 1998), because the few organisms present may have the appropriate sensitivity. The ASPT score may be lower where, for example, a sand bed river in pristine condition may produce a low ASPT, as it will be occupied by hardy, adaptable taxa. Chutter (1998) also pointed out that ASPT was a more reliable measure of the health of good quality rivers than that given by the SASS score (Dickens & Graham, 2002).

There has been much debate, especially relating to aquatic systems, as to what taxonomic level (either family- or species-level) is most suitable for biological monitoring (Guèrold, 2000; Lenat & Resh, 2001). The consensus is that whenever possible, sorting to species level is advisable. However, in some conditions, sorting to family may also be acceptable (Baily *et al.,* 2001; Lenat & Resh, 2001). Not surprisingly, analyses using higher taxa appear to be better suited to studies at broader geographic scales (Hewlett, 2000). A study by Basset *et al.* (2004) to evaluate this problem indicated that among higher taxa (orders, families and guilds) the data set with the highest discriminant power was at the family level, with or without rare taxa. They also noted that to include more taxa in the data sets does not necessarily ensure that these data sets will have greater discriminatory power. The quality and choice of the information included in the data sets are more important in this regard (Basset *et al.*, 2004).

Considering the above, SASS5 can be seen as a suitable biomonitoring system that not only providing a rapid assessment, but it is also cost effective and easily managed by non-professionals. Taking in consideration that ASPT is a less variable measure than the SASS score or the Number of Taxa. SASS should be the obvious index of choice. However, in some instances, particularly in polluted water, the SASS score becomes more meaningful (Chutter, 1998; Dickens & Graham, 2002).

The issue relating other factors influencing the insect presence as well as their numbers is one that needs to be kept in consideration when assessing, but as the study by Zomora-Munoz *et al.* (1995) on the BMWP system indicated that the relation to seasonality (including all physical factors of the river) and ASPT is significant, but the relationship between pollution and ASPT was even greater.

5.3 HUMAN MANIPULATION OF THE RIVERINE SYSTEM

5.3.1 Influences of impoundment on the riverine system

A typical stream network resembles a tree, with a large, widely spread-out root-like base eventually leading to a single, major trunk. The stream water body exhibits universal coherence and physical continuity, and naturally there are also regular or even continuous (Vannote *et al.*, 1980) longitudinal changes of ecological conditions (Zwick, 1992). Some of these expected changes include the broadening of the river as the system gains water from the catchment, influencing the amount of canopy cover along the system, and thus indirectly influencing the food sources in the system. More sunlight means more algal growth, while less canopy cover to water ratio reduces the amount of debris in the system. More sunlight also increases water temperature (Davies & Day, 1998). The mountain river zone also normally has high flowing areas with an array of different habitats from waterfalls to back water pools. With the amount of habitat diversity expected to be high, invertebrates have to be highly adapted for the harsh conditions (Zwick, 1992), and therefore diversity and number of species per insect family is supposed to be the highest in this zone (Compin & Cereghino, 2003).

As the gradient difference drops in the upper river zone, the river gets calmer and is seen as the intermediate between two extremes and therefore having the highest diversity as well as species richness (Davies & Day, 1998).

Stream flow in the lower river slows down as it reaches the low laying areas. Diversity also drops due to lack of food as very little debris is deposited in this area and the amount of silt in the water prevents algal growth.

Anthropogenic control over the flow of running water, usually by means of dams and reservoirs, has influenced nearly all of the world's major river systems (Williams & Feltmate, 1992).

Dams vary widely in size, purpose and operation, and this influences their impact on river ecosystems (Petts, 1984). Dams also differ in whether water is released from the surface or near the bottom of the dam, or both (Allan, 1995). Furthermore, the purpose of the dam implies different effects that will impact on the riverine system. Dams constructed for irrigation must store as much water as possible during the rainy season for subsequent release during the agricultural growing season. Flood control reservoirs maintain only a small permanent pool in order to maximize storage capacity. Navigation requires water storage in upper reaches to offset seasonal low-flow conditions, and may be complemented by a system of locks and dams (Allan, 1995). Hydroelectric dams store water for release to meet regional energy demands, which can vary seasonally or over the course of one day. Hydroelectric dams can differ substantially in their operation. “Run-of-the-river” dams release water at the same rate it enters the reservoir and usually are of low height and are considered to have relatively minor adverse effects. “Peaking” hydropower dams meet daily fluctuations in energy demand by allowing water to flow through turbines only at certain times, usually from mid-morning through early evening, and are considered to seriously affect aquatic life (Allan, 1995).

Unquestionably impoundments cause fundamental changes in community structure and ecosystems functioning as a naturally free-flowing and continuous river course is transformed into river segments interrupted by impoundments (Allan, 1995).

The effects of impoundments include a range of changes in the physical conditions downstream of the dam, especially modification of the flow and temperature regimes, and usually greater water clarity. Water quality changes may be slight or considerable, depending on water surface or whether deep water is released. The modified physical and chemical conditions result in changes in the plant and animal life of the river. In addition, reservoirs impede the downstream passage of young migrating fishes, while dams are barrier to upstream migrations. More indirectly, dams and especially a series of dams, break the upstream–downstream connectivity, a natural feature of rivers (Allan, 1995).

Furthermore, the presence on an impoundment obviously changes the discharge and current for some distance downstream, and a series of large dams will completely alter a river's natural periodicity of flow. In addition to this dampening of flow variability, transbasin diversions and evaporative losses can result in an overall reduction in discharge (Allan, 1995). Transport of suspended particles and the amount of fine sediment on the streambed are affected by the presence of the reservoir and the river's altered flow regime (Barrow, 1987).

5.3.1.1 Sediment levels

In the case of an impounded river, the concentration of sediment, dissolved salts, and other parameters released depend on the release pattern of the dam, bottom or surface, and can affect aquatic organisms for up to 200 km downstream from the source (O'Keeffe *et al*., 1990; Palmer & O'Keeffe, 1990; Madikizela, 2001).

This increase in sediment or silt causes an avoidance of sedimented regions due to the loss of interstitial space between stones, with behavioural observations revealing that the insects will not excavate fine particles (Williams & Feltmate, 1992).

5.3.1.2 Temperature

Impoundment will alter a river's temperature regime to varying degrees, more strongly so in the case of large reservoirs with deep release dams located on temperate rivers. Because the large volume of a reservoir has considerable thermal inertia, diel temperature fluctuations are reduced or eliminated. Deep release reservoirs may also reduce the extent of seasonal temperature variation. Provided the reservoir has sufficient depth and residence time, thermal stratification will occur during warm months, in which the epilimnion, or surface waters, are much warmer than the hypolimnion, or deep waters (Stanford & Ward, 1989). Changes in temperature regime of the magnitude found below deep release dams have a significant impact on the benthic fauna. A reduction in species richness is likely for several reasons. Warmer than normal winter temperatures eliminate the thermal cues needed by many species to break egg diapause. Cool summer temperature can also have an adverse effect, because there are too few degree days to complete development, life cycles loose their synchrony, or due to changes in the temporal pattern of growth and development. The mechanisms by which altered temperature influences invertebrates clearly warrants further study, but unquestionably the total effect can be profound (Allan, 1995).

5.3.1.3 Dissolved oxygen

By far the most serious change in water quality is due to the release of oxygen-depleted water from the hypolimnion of a deep reservoir. However, turbulence usually re-oxygenates the water within a short distance, and artificial aeration is a relatively simple solution (Allan, 1995).

5.3.1.4 Water clarity

Downstream changes in water quality depend on the limnological processes within the reservoir and depth of water release. Deep release dams generally cause the most adverse effects. Water clarity typically increases because of the reduced sediment in transport, often with significant effects on plants life (Allan, 1995). Whenever dams cause enhanced water clarity and reduced variability of stream flow, there is usually a greater abundance of periphyton of higher plants than is found elsewhere in the river (Stanford & Ward, 1979).

5.3.1.5 Invertebrates

The building of dams imposes a lentic habitat within a lotic system (Brittain & Saltveit, 1989; Mackie, 1998). The aquatic communities must suddenly adjust to the changes in physical, chemical and biological attributes of riverine systems to those of lacustrine systems (Armitage, 1990). Some species are adapted to a lotic existence and perish when a lentic system is imposed upon them. Others, mostly highly tolerant forms like Chironomidae, exploit the new habitat and increase in biomass. This is the case for Ephemeroptera, Trichoptera and Plecoptera following the formation of an impoundment, which disappeared almost immediately, but were replaced by high densities of Diptera (Williams & Feltmate, 1992). Field assessment on the distribution and diversity of the aquatic Coleoptera below and above the Hely-Hutchinson Reservoir, Table Mountain, Cape Town, South Africa, indicated a negative effect of the reservoir on the aquatic fauna. The presence of the reservoir dramatically reduced (73%), the existence of certain Cape endemic and internationally threatened species of Elmidae, Dryopidae and Hydraenidae (Turner, 2000).

If impoundments were constructed on streams of stream order 3, 4 or 5, species diversity usually (depending on depth and size of impoundment) declines, but if it does not, the species assemblage certainly changes from one dominated by shredders and lotic filter feeders, grazers and predators to that of herbivores and lentic filter feeders and predators (Williams & Feltmate, 1992).

5.3.1.6 Fish

It is well known that hydrological and ecological changes associated with impoundments have contributed to the loss or reduction of population of migratory fishes and are cause for serious concern, not only because of the economical value of fish, but also for their contribution to regional biodiversity and their significance to local and regional cultures (Li *et al.*, 1987; Pringle *et al.,* 2000; Reid, 2004). Habitat changes, such as altered downstream water temperatures and the creation of reservoir lakes, also favour the invasion or introduction of alien species that can further adversely affect indigenous fish populations (Quinn & Kwak, 2003).

5.3.2 Influences of the Klein Plaas dam

The Mountain stream zone has remained relatively unaffected (Brown & Dallas 1995), with the riparian belt still fully intact. The area below the dam shows mainly indigenous riparian vegetation, but there is also greater abundance of alien species. This may be due to the reduction of the water level, creating an opportunity for invasive aliens to settle before the indigenous riparian vegetation can move in. The upper river zone is also being affected by agricultural runoff, particularly during the summer months (Brown & Dallas 1995).

The SASS5 as well as the ASPT scores indicated the dam to be a clear disturbance factor in the ecosystem because downstream had SASS5 values half that of upstream. The normal degration of water quality between the White bridge station and the Concrete bridge station was very small in comparison to the two sampling areas divided by the dam. Simpson’s index of diversity indicated greater diversity levels high up in the system and not downstream. While all the physical data indicated significant differences between the upstream and downstream areas with the exception of Dissolved Oxygen.

Furthermore, there was a clear difference in the flow speed downstream, an increase in sediment and silt into the system due to the deep-water release, as well as an increase in algal growth downstream.

5.3.3 Trout farms

Studies done on the effect of trout farms in the past as well as this study indicated that Ephemeroptera, Plecoptera and Trichoptera taxa richness was significantly less just below the outfall compared to stream sites above the intakes of the farms, indicating reduced water quality (Loch *et al*., 1996). Furthermore, literature indicates that pollution-tolerant taxa like Chironomidae, Simuliidae, Oligochaeta and Sphaeriidae have higher abundances just below the outfalls (Loch *et al*., 1996) as was also observed in this study, indicating that the trout farm might add to the effects that the reservoir have on the system.

5.3.4 Inter-basin transfer

Inter-basin water transfers form a major form of river basin manipulation. In South Africa they are increasingly being used to reconcile the problems of water distribution within the country (Snaddon *et al*., 1998).

It is obvious that interfering with the flow and water levels of a riverine system will in the end manipulate the ecology of the biota, even more so if water from different sections of the stream are mixed, bringing with it altered physical features as well as distributes biota from other riverine systems.

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Chapter six

CONCLUSIONS

Healthy, effective functioning rivers provide a wealth of reliable benefits to people, from the provision of good quality drinking water, to resources such as fish and reeds, to recreational pleasure. Poorly functioning river systems gradually lose their valued attributes, require continual expensive remedial actions, or are costly to the nation in other ways, such as through collapsing banks, sediment-filled dams and water quality problems. It is therefore important for government, but more so for humans to be able to evaluate their influences on the riverine system. Bioassessments and monitoring was thus developed for the rapid evaluation of these systems, using organisms of both fauna and flora as an indication of the ecological state of the system. The most valuable information for prediction, monitoring and assessment is based on understanding why changes will occur or have occurred, because this allows informed judgements and decisions to be made about present or future impacts.

In an ideal situation the quality of running water should be assessed by the use of physical, chemical and biological parameters in order to provide a complete spectrum of information for appropriate water management. However, such a study needs much more time and expenses than the study of the biological parameters alone.

Bioindicator based studies must be simple and easily repeated by different people in various situations, be feasible in different environments, and be suitable for assessing large areas. Therefore, the South African Scoring System (Version 5) is seen as a suitable evaluation format that fall in these categories, but there is concern that other factors like shade, climate, water level, among others, may influence the distribution as well as abundance of invertebrates present in the system that SASS5 evaluates. This is generally agreed to be the case, but overall toxicity as well as the degration of water quality indicate greater influence on distribution and abundance than any other parameters. SASS5 indicates problematic areas with changes in water quality. These indications can then be studied further to determine the point source of the impact responsible for the degration in the water quality.

Improvements recommended for SASS5 evaluation is to incorporate a predictable phase that can be correlated to lesser climatical as well as vegetational regions and the zonation of the river itself. Other studies also take rarity in consideration and into the equation and might indicate a significant difference if implemented.

This study furthermore indicated that the Klein Plaas dam as an impounding body of water, along with its activities from the inter-basin water transfer project as well as the University of Stellenbosch Experimental cage Trout farm, influences the riverine system of the Eerste River negatively. All factors ranging from diversity, present species, SASS5 evaluation to the physical measures indicate significant changes in the water quality and are also visible in the surrounding vegetation.

Successful application and use of biomonitoring data are only possible when the biological and behavioural responses of invertebrates subjected to natural and anthropogenic perturbation are understood. Further studies are thus needed focusing on the morphology of the species present in the system, providing identification to species level, study of the ecology and behaviour of the invertebrates as well as the impacts of significant factors like for example, shade, water depth, temperature, conductivity, pH and insecticides, on the distribution and abundance of benthic macroinvertebrates. There is also a need for development of procedures that can be applied to larger rivers or rivers in flood, as well as lakes and impoundments.

Humans may have the ability to manipulate the environment to suit their needs, but this requires a responsible approach. Our present generation must therefore stand up and be accountable for our actions, focusing our knowledge and intuition toward a better future that includes the availability of clean, freshwater for all the nations of the world.