The impact of alien invasive smallmouth bass (*Micropterus dolomieu*) on the indigenous fish of the Rondegat River: A quantitative assessment with implications for rehabilitation

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ABSTRACT

Alien invasive fishes are a growing concern in inland water ecosystems around the world, as they are capable of causing serious damage, especially to indigenous fish populations. Mechanisms include direct predation on indigenous fish by alien predatory species, competition for food and space between native and introduced species, the introduction of alien parasites and pathogens, and general environmental degradation. The Cape Floristic Region (CFR) of South Africa, which is defined by a unique and highly diverse floral kingdom, is also home to a unique and highly threatened ichthyofauna. This ichthyofauna consists of relatively few species, but exhibits the highest proportion of endemicity in the country. Threats to fishes include habitat destruction through bulldozing and water extraction, water quality degradation, restriction of migration by dams and weirs, and alien invasive fishes. The North American smallmouth bass (*Micropterus dolomieu* Lacepede) has long been regarded by nature conservators as the most threatening invasive species, and much anecdotal and survey data suggests that this species has had a major role in the depletion and extirpation of indigenous fish populations throughout the CFR. However, no study has ever properly quantified this impact relative to other potential threats that could have precipitated the perceived indigenous fish losses. This is a problem, as it makes the implementation of active control measures difficult to justify to a skeptical public and potential sponsors.

The Rondegat River is a tributary of the Olifants River, which rises in the Cedarberg Mountains and flows into the Clanwilliam Dam reservoir. It is partially invaded by *M. dolomieu*, which have penetrated the lower quarter of the river up to a waterfall barrier. This river is home to five species of indigenous fish, including the Clanwilliam yellowfish (*Labeobarbus capensis* Smith), Clanwilliam redfin (*Barbus calidus* Barnard), fiery redfin (*Pseudobarbus phlegethon* Barnard), Clanwilliam rock catfish (*Austroglanis gilli* Barnard) and the Cape galaxiid (*Galaxias zebratus* Castelnau). This project was designed to quantify the impact of *M. dolomieu* on these species relative to the alternate potential impacts of physical habitat degradation from agriculture and alien invasive riparian trees, and of food availability.

Seasonal surveys were conducted at eight sites on the river in September, October and November 2003, and in April 2004. Four sets of riffles and pools were selected upstream of the waterfall barrier, and four below. Quantitative electrofishing was used to survey fish in riffles, while snorkeling surveys were conducted in pools. All fish species were also caught with seine and fyke nets for dietary analyses. Physical habitat variables were measured at each site, and used to assess changing habitat quality between the sites. Invertebrate samples were also taken along with visual abundance estimations of other food types, to gauge food availability.

Fish surveys revealed the loss of *B. calidus*, *P. phlegethon*, *A. gilli* and *G. zebratus* at bass-invaded sites. *Galaxias zebratus* was only found in the upper reaches of the river, and so was possibly never common in the lower river. *Labeobarbus capensis*, while still below the waterfall, appeared to be suffering from near-total loss of post-spawning recruits. Discriminant function analyses revealed sedimentation to be a key factor of habitat degradation that characterized invaded sites. However, linear regressions between habitat variables and indigenous fish densities indicated sedimentation to not be a significant negative influence on indigenous fish distributions. Although sedimentation did not appear to influence *A. gilli* densities at non-invaded sites, it is highly likely that it increased the vulnerability of this species to *M. dolomieu* in the lower river, by removing benthic cover used to avoid predation. In the case of all species, food availability did not seem an important factor in dictating fish distributions. Consequently, predation by *M. dolomieu* was confirmed as the critical mechanism behind the loss of *B. calidus*, *P. phlegethon*, *A.gilli* and juvenile *L. capensis* in the lower river.

A rehabilitation plan is proposed for the Rondegat River. Central to this plan is the formation of a conservancy between the land-owners of the catchment and the custodians of the Cedarberg Wilderness Area. The most effective strategy will be to eradicate *M. dolomieu* from the lower river with piscicides, while at the same time taking steps to rehabilitate the riparian zone throughout the river. A holistic rehabilitation programme such as this would provide an excellent model for future rehabilitation efforts within the CFR.

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CHAPTER 1: INTRODUCTION

1.1: THE GLOBAL THREAT OF BIOLOGICAL INVASIONS

The spread of alien invasive species is now recognised one of the greatest threats to the ecological well-being of the planet (Matthews & Brand, 2004). The impacts of these organisms on biodiversity are considered to be second only to that of human population growth and its associated activities (Pimental, 2002). Invasive species are considered to be the leading cause for global extinctions in birds, and the second-most important factor in the extinctions native fishes in the United States (Clavero & Garcia-Berthou, 2005).

An invasive species can be defined as a non-native species that, having established populations in a new habitat, are able to expand their range within that habitat (Coulatti & McIsaac, 2004). Today, both aquatic and terrestrial ecosystems have been invaded by alien vertebrates, invertebrates, plants and micro-organisms, many of which have had a catastrophic effect on the native species within those ecosystems.

Biological invasions are, for the most part, mediated by humans. Marine invertebrates are able to cross entire oceans in the bilge of ships (Carlton, 1999), while many other species "hitchhike" on planes, trucks and shipping containers, or are carried in amongst logs, fruits, seeds and vegetables (McNeely, 1999). Many species have, however, been introduced intentionally by humans, generally for economic purposes (NcNeely, 1999; Mack *et al.*, 2000).

The impacts of alien invasive species are wide ranging and profound. Invasive pathogens can affect the health of indigenous species, as well as humans (Mack *et al.*, 2000; Lobo, 2002). In the case of plants and animals, invaders can hybridise with indigenous species, diluting their genetic stock (Mack *et al.*, 2000). Invasive animals can also lead to the decline, and eventually the extinction of native species, directly through predation or competition (McDowall, 2003) or indirectly through the transformation of landscapes, resulting in the loss of habitat for indigenous species (Matthews & Brand, 2004).

Increasingly governments are beginning to realise the threat posed by alien invasive species, and have developed policies to prevent the spread of new alien species (Bean, 1999). Furthermore, the rise in biocontrol agents in the form of species-specific pathogens and parasites being introduced to control the invasive species, are in the case of several plant species, beginning to turn the tide on the invasives (Impson *et al.*, 2004).

Biocontrol agents are, however, much harder to find for invasive vertebrates, and there is still a substantial knowledge gap as to how to manage those species that are not only spreading, but doing so to the detriment of the environment they have invaded. Only by understanding the mechanisms by which these invasive species affect native biota, can scientists and other stakeholders hope to mitigate against and ultimately nullify their impacts.

1.2: THE IMPLICATIONS OF FISH INVASIONS FOR NATIVE FISHES

1.2.1: The translocation of freshwater fishes: an historical perspective

Freshwater ecosystems are among the most invaded in the world, with introductions of alien species continuing to occur at a high rate (Moyle, 1999). Invasions in aquatic ecosystems occur as a result of official introductions, intended to aid human interests, intentional but illegal introductions to serve a private purpose, accidentally as a result of human transport, or naturally without the aid of human intervention (Townsend, 1996). While many recent introductions of aquatic organisms have been unintentional, the introduction of fish species historically has mostly been deliberate. The first recorded translocations of fishes were performed by the ancient Romans, who introduced such species as the common carp (*Cyprinus carpio* Linnaeus) to various countries in Europe for stocking ornamental ponds (Holcik, 1991). In the 1600s the goldfish (*Carassius auratus* Linnaeus) was introduced to North America (Crossman, 1991), presumably also for ornamental purposes. In the nineteenth century, the global translocation of fish species rapidly accelerated, with species from Europe being introduced into Africa (de Moor & Bruton, 1988), North America (Crossman, 1991) and Oceania (Townsend, 1996). Much of this increase was given momentum by the rapid development of aquaculture practices, such as the artificial fertilization and hatching of salmonids, as well as increased infrastructure to aid successful transportation (Krueger & May, 1991).

In the 1950s and '60s, a second, more rapid increase in fish introductions occurred (Holcik, 1991; Crossman, 1991). This was driven by an increase in government-sponsored stocking programmes, with national authorities seeking to improve the fish stocks of their inland waters. In the case of large lakes, this followed a need to supplement or replace failing indigenous fisheries, which were suffering from over-exploitation and human-mediated habitat degradation (Moyle, 1999; Ogutu-Ohwayo, 1993), although in some instances, such as the stocking of artificial lakes created by dams, the purpose of the introduction was to create a lacustrine fishery where none had existed before (Fernando, 1991; Ogutu-Ohwayo & Hecky, 1991; Hall & Mills, 2000). In river systems, the purpose of fish introductions was often to supplement or create local sport fishing opportunities (de Moor & Bruton, 1988). The post-war increase in fish invasions was also characterised by a diversification in the mechanisms of introductions. The number of accidental invasions increased, a result of intensified utilisation and modification of inland water environments (Hall & Mills, 2000). Other releases of exotic fishes resulted from escapes and the illegal dumping of aquarium species (Moyle, 1999), the transfer and dumping of larvae in ship ballast water (Carlton, 1999), escapes of aquaculture fish from farms into adjacent waters (Krueger & May, 1991), the dumping of live bait species (Dextrase, 1996), as well as the illegal translocation of angling species by anonymous individuals from their original water bodies into nearby rivers and lakes (Moyle, 1999; Hall & Mills, 2000), presumably to enhance the recreational potential of the receiving bodies.

These forms of unofficial introductions indicate a level of ignorance or disregard for the environment in members of the general public (Moyle, 1999), a problem that conservation authorities and environmentalists throughout the world must combat on a daily basis. However, even in the official introductions of freshwater fish species, insufficient thought has historically been given by the authorities to the effect they would have on the environment, or to what the long-term repercussions would be if the introduced fishes became invasive (Moyle, 1999). Today, the awareness of fish invasions and their consequences is growing, but much research is still needed to understand the consequences that might be expected.

1.2.2: Impacts of alien fish on indigenous fish fauna

The introduction of fishes into freshwater environments can have many profound consequences to the local ecosystem. Alien fish have been known to negatively affect amphibian (Adams, 2000; Gillespie, 2001) and invertebrate (Marshall, 1991) diversity. Many introductions directly affect the biodiversity of native fishes. In the lakes of the north-eastern United States of America, statistical analysis has shown that the number of introduced fish species present can be used to predict the indigenous fish species richness in a lake, with more aliens invariably correlating with fewer natives (Whittier & Kincaid, 1999). This is a global phenomenon that is precipitated by several mechanisms, not all of which lead directly to the loss of a species, but nearly all of which have a negative impact to some degree.

(a) Parasite and disease transfer

An immediate consequence of introducing an exotic fish species into a freshwater ecosystem is that it may be carrying parasites or diseases that could be transferred onto indigenous species. These pathogens could seriously disrupt the local ecosystem, by reducing the size or competitive ability of the native populations. An example of this phenomenon is the introduction of whirling disease with European brown trout (*Salmo trutta* Linnaeus) into the U.S.A.; a pathogen lethal to indigenous trout, the disease is credited with allowing the brown trout to easily establish itself in the rivers it was introduced into (Moyle, 1999). In southern Africa, common carp have been implicated in the introduction of several fish parasites, some of which have since spread to indigenous fishes. The cestode *Bothriocephalus acheilognathi* (Yamaguti) was introduced with grass carp (*Ctenopharyngodon idella* Valenciennes), and has infected a wide range of indigenous cyprinids (Bruton & Van As, 1986). The problem with pathogen transfers is that they are seldom detected (Fernando, 1991; Stewart, 1991) and are therefore very difficult to manage. The identification and control of these organisms is a difficult undertaking, but essential to the conservation of indigenous fish around the world.

(b) Predation

Perhaps the most profound effect that invasive fish species have had on indigenous fish biodiversity is reduction of numbers through direct predation. Some of the most spectacular losses of freshwater fish species have resulted from the introduction and establishment of exotic predators, usually sport fishes introduced to enhance fisheries or recreation (Ogutu-Ohwayo & Hecky, 1991). Probably the most publicised example of impacts of an introduced predator on biodiversity of indigenous fish comes from Lake Victoria, in East Africa. Up until the 1950s, the lake was home to the largest known species flock of fishes in the world. Up to 300 species of haplochromine cichlids resided in the lake, all having speciated within the lake in the recent geological past (Goldschmidt *et al.*, 1993). In the late 1950s, the Nile perch (*Lates niloticus* Linnaeus) was introduced by British colonial authorities in order to improve production in the local fishery (Baskin, 1992). Although the Nile perch had an insignificant impact on the lake at first, in the 1980s the population began to increase exponentially, causing the decline through predation of hundreds of species of haplochromines, several of which became extinct (Oguto-Ohwayo & Hecky, 1991; Baskin, 1992). This phenomenon has been repeated in two Ugandan lakes: Lake Kyoga, where haplochromines are today virtually absent (Oguto-Ohwayo & Hecky, 1991) and Lake Nabugabo where the number of native species of cichlids has been reduced by half (Oguto-Ohwayo, 1993). In southern Africa, introduced sport fish such as the North American black basses (*Micropterus* spp.) have had measurable impacts on indigenous fish biodiversity. In the upper Manyame River, Zimbabwe, the presence of introduced largemouth bass (*Micropterus salmoides* Lacepede) corresponded with a 99% decrease in numbers of small native *Barbus* spp. (Gratwicke & Marshall, 2001). In the Blindekloof River, South Africa, an invasion of largemouth bass resulted in three endemic fish species temporarily vanishing from the invaded reaches, before the removal of the bass allowed their recovery (Skelton, 1993). Rainbow trout (*Oncorhynchus mykiss* Walbaum) have also been implicated in the local extinctions of eight species of minnow (*Barbus* spp.) and the southern kneria (*Kneria auriculata* Pellegrin) in South African streams (Bruton & Van As, 1986).

(c) Hybridization

Apart from the predatory threats posed by introduced fishes to their native counterparts, in the case of closely related species, the invaders may pose a genetic threat as well. The ability of salmonids to hybridise is well documented. In western North America, the ubiquitously introduced *O. mykiss* has had a substantial impact on the gene pools of the native golden trout (*Oncorhynchus aquabonita* Jordan) and various strains of cutthroat trout (*Oncorhynchus clarki* Richardson). Interbreeding has resulted in the formation of hybrid populations, which in some areas are replacing the native stock (Krueger & May, 1991). In Europe, where the brown trout has been introduced widely, it has had a severe impact on the marbled trout (*Salmo marmoratus* Cuvier). A 1983–1985 survey in Slovenia found only one river left that contained a pure stock of marbled trout. All other rivers had greatly reduced populations, of which 0.8–51% of individuals were hybrids (Crivelli, 1995). In Africa, the widespread translocation of tilapiine cichlids has resulted in several cases of hybridisation. In parts of the Limpopo River, South Africa, the native Mozambique tilapia (*Oreochromis mossambicus* Peters), has begun hybridising with introduced Nile tilapia (*Oreochromis niloticus* Linnaeus) and may in time be completely replaced in the system by hybrids (van der Waal & Bills, 2000).

Apart from the diluting effect that hybridisation can have on indigenous fish diversity, hybridisation between exotic species can lead to further problems. In Australia, interbreeding between various varieties of common carp (*C. carpio*) in the wild has resulted in the appearance of an aggressively invasive strain known as the "Boolara carp". More hardy and vigorous than any original strains, the Boolara strain has become a difficult management issue in the Murray-Darling River system (Arthington, 1991).

(d) Competition

As an impact, competition can be hard to prove, as while the mechanisms are generally well understood, the results of competitive interactions are not often effectively demonstrated (Crivelli, 1995). A working definition of competition is that it is "an interaction between individuals, brought about by a shared requirement for a resource in limited supply, and leading to a reduction in the survivorship, growth and/or reproduction of at least some of the competing individuals concerned" (Begon *et al.*, 1996). Thus, the increased pressures placed on a native species by competitive interactions with an introduced species, whether for food or habitat, can lead to the native species' demise. In streams, competition may fluctuate seasonally, becoming more intense in the dry season when there is less habitat and food as a result of reduced flow (Krueger & May, 1991). Arthington *et al.* (1989) noted that disturbance might greatly increase the competitive effects of alien species on indigenous species by restricting their resources through habitat loss, making them more vulnerable to strong competitors.

Interspecific competition between native and introduced salmonids is believed to have taken place throughout North America. Rainbow trout appear to have replaced the native brook trout (*Salvelinus fontinalus* Mitchill) in the streams of the Appalachian Mountains as a result of being able to better exploit the shallow riffle habitat characteristic of those streams (Krueger & May, 1991). In Europe, the mosquitofish (*Gambusia affinis* Baird & Girard) is cited as competing for habitat with certain fish species of the family Cyprinodontidae, one of which has been displaced into more saline habitats when *G. affinis* is present (Crivelli, 1995). Competition can also be size- and age-specific. In Lake Trasimeno, in Italy, largemouth bass are believed to out-compete juvenile pike (*Esox lucius* Linnaeus) for food in winter, resulting in increased juvenile mortality in the pike (Lorenzoni *et al.*, 2002). In South Africa, banded tilapia (*Tilapia sparmanni* Smith) and bluegill (*Lepomis macrochirus* Rafinesque) are thought to compete with indigenous species for food (de Moore & Bruton, 1988).

(e) Perceived beneficial effects

Sometimes, the introduction of an alien fish into a water body can lead to a temporary benefit for certain native species, especially when the alien is a prey species. In Lake Kariba, for example, the introduction of Tanganyika sardine (*Limnothrissa miodon* Boulenger) led to a short-lived population explosion in the lake's indigenous top predator, the commercially important tigerfish (*Hydrocynus forskalii* Cuvier) (Marshall, 1991). In reservoirs in several Asian countries, the introduction of African tilapiines has apparently increased the numbers of indigenous cyprinids caught each year, thereby substantially improving local fishing productivity (Fernando, 1991). The tilapiines do not appear to compete with the indigenous species, but rather increase the eutrophication of the water body, providing more food in the way of algae for the cyprinids, most of which are omnivorous (Fernando, 1991).

When looking at the potential benefits of such introduction, one must bear in mind that what is considered an unqualified success by fisheries managers may not be good for the natural biodiversity. In the case of the Asian tilapia introductions, the fact that the indigenous fishery improved does not necessarily mean that the reservoir's ecosystem benefited. It is possible that the introductions had a negative effect on the rest of the lake's native fish and invertebrate assemblages, and may have actually decreased the overall local biodiversity. When assessing the impacts of introduced fishes, it is important that one looks beyond those species that the managers consider commercially valuable, and that proper consideration of all species that make up the indigenous community is taken when deciding on whether or not to introduce a fish species.

1.3: THE CONTEXT OF THIS DISSERTATION: FISH CONSERVATION IN THE CAPE FLORISTIC REGION

1.3.1: The Ichthyofauna of the Cape Floristic Region

The freshwater fishes of South Africa can be divided into two distinct faunas: the tropical Zambezian ichthyofauna, and the southern temperate ichthyofauna. The southern temperate ichthyofauna (STI) is not very diverse (33 species) but contains many endemic species (Skelton *et al*, 1995). Perhaps the most important region pertaining to the conservation of the STI is the Cape Floristic Region (CFR), which encapsulates the Cape Floral Kingdom, one of the world's unique and endangered bioregions (CEPF, 2002). The rivers of this region are notable for their endemic fish fauna, which form a unique "Cape" component of the STI (Skelton, 1983), and which exist in highly restricted ranges within the CFR's river systems (Skelton, 1987). Although relatively low in species richness, the Cape fish fauna is arguably the region's most endangered component, (Impson *et al.*, 1999). At the most recent assessment, of the 19 species indigenous to the region, 16 are endemic, while 15 are listed as threatened (Impson *et al.*, 2002).

The Cape ichthyofauna is dominated by species of the family Cyprinidae (Impson *et al.*, 2002). They include species such as the redfin minnows, a very attractive group of fishes that are typically found in clear perennial mountain streams. The largest of the Cape cyprinids is the Clanwilliam yellowfish (*Labeobarbus capensis* Smith), which can grow to nearly a meter in length (Skelton, 2001). Other families found in the CFR include the Anabantidae with 1 species, the Austroglanididae with 2 species and the Galaxiidae (Impson *et al.*, 2002). There is currently only one species of galaxiid recognised, the Cape galaxias (*Galaxias zebratus*). Recently completed genetic work however, has indicated there to be at least six species of galaxiid present, though these are still in the process of being confirmed and described (Swartz, 2003). An important feature of all of these species is that they generally exist within highly restricted natural ranges, with some species such as the Twee River redfin (*Barbus erubescens* Skelton) being native to a single tributary of the Olifants River (Skelton, 1987).

The conservation of all these fish species is becoming a critical issue in the overall conservation of the CFR, and the key to conserving them effectively will ultimately be to understand and mitigate against the various threats that currently endanger them.

1.3.2: Alien fish introductions in the CFR

Today, the presence of alien invasive fishes is considered a major threat to indigenous species in Western Cape rivers (Skelton, 1987; Impson & Hamman, 2000). Invasive species currently include brown trout (*S. trutta*), rainbow trout (*O. mykiss*), largemouth bass (*M. salmoides*), smallmouth bass (*Micropterus dolomieu* Lacepede), bluegill sunfish (*L. macrochirus*) common carp (*C. carpio*), grass carp (*C. idella*), mosquitofish (*G. affinis*) as well as translocated southern African native fishes, such as the Mozambique tilapia (*O. mossambicus*), banded tilapia (*T. sparrmanii*) and sharptooth catfish (*Clarias gariepinus* Burchell) (Gaigher *et al.*, 1980; Skelton 2001). Other alien species, such as spotted bass (*Micropterus punctulatus* Rafinesque) and brook charr (*Salvelinus fontinalis*

Mitchill) have been introduced but apparently have failed to establish healthy populations (Skelton, 2001).

Of these species, the smallmouth bass (*M. dolomieu*) appears to have caused the most damage to the indigenous fish populations (Gaigher, 1973; Hamman & Jordaan, 1988). Introduced in 1937 for angling purposes, *M. dolomieu* now inhabits the Berg, Breede and Olifants river systems in the CFR (de Moor & Bruton, 1988). It can adapt to most riverine habitats, including both acidic and warm waters (de Moor & Bruton, 1988), which means it can survive in most of the riverine environments of the CFR.

Since its introduction, *M. dolomieu* has been implicated in the disappearance of six fish species from sections of the Olifants River (de Moor & Bruton, 1988), while in the Berg and Breede Rivers, it is associated with the disappearance of local endemic cyprinid species and the Cape kurper (*Sandelia capensis* Cuvier) (de Moor & Bruton, 1988). In the Olifants-Doring river system, *M. dolomieu* are considered by many conservationists past and present to be one of the biggest threats to indigenous species (de Moor & Bruton, 1988). In 1949 Thomas Brooks, one of the men responsible for the introductions, wrote that he noticed that the yellowfish fingerlings that were once so common in the mainstem in summer had almost vanished (de Moor & Bruton, 1988). By 1973, two endemic cyprinids, the Clanwilliam redfin (*Barbus calidus* Barnard) and the fiery redfin (*Pseudobarbus phlegethon* Barnard), were no longer found in the mainstem of the Olifants (Gaigher, 1973). After carrying out a fish survey on the Olifants River for the Cape Department of Nature Conservation, Gaigher (1973) suggested that indigenous fishes were only able to survive in the system above natural barriers where bass could not penetrate. He also surmised from the survey that another endemic species, the Clanwilliam rock catfish (*Austroglanis gilli* Barnard), was now restricted in its range by the presence of "exotic predators" (Gaigher 1973).

While these surveys all produced compelling circumstantial evidence, the researchers involved did not measure the impact of bass relative to other potential causes for the shrinking distributions of endemic fish in the system.

1.3.3: Other direct threats to indigenous fishes in the CFR

Impson and Hamman (2000) stated that apart from the impact of alien invasive fishes, the greatest threat to indigenous fishes in the Western Cape is habitat loss. The major cause of habitat loss and degradation in the region is agriculture (Gaigher *et al.*, 1980; Impson & Hamman, 2000). Perhaps the most obvious threat posed by agriculture to fish in the C.F.R. is that of water extraction. Because the CFR is a winter rainfall area, the dry, hot summer coincides with the time of greatest water need for agricultural purposes (Gaigher *et al.*, 1980). As a result, the rivers of the region have come under increasing stress from water extraction during the summer months. In the Olifants River for example, unregulated water extraction now causes entire reaches of the river between Citrusdal and Clanwilliam to dry up during the summer months, making this segment of the river highly inhospitable to fish (pers. obs., 2004). Dr Paul Skelton (quoted in Gaigher *et al.*, 1980) surmised that water extraction was probably causing the decline of six threatened Cape species.

Siltation, from inappropriate land-use, is also regarded as a negative influence on fish. Gaigher *et al.* (1980) noted it as a threat to instream invertebrates, which are an important food source for some indigenous fish. Bills (1999) noted sedimentation as especially damaging to endemic benthic catfish of the Family Austroglanididae, because it removed benthic cover from the riverbed, increasing their vulnerability to predation by alien fish. Another major threat to indigenous fish species in the CFR is that of eutrophication, as a result of high-nutrient agricultural runoff (Gaigher *et al.*, 1980).

Because most of the rivers in the CFR are naturally oligotrophic, meaning that they have very low natural nutrient contents (King *et al.*, 1979), the fish native to these rivers are not adapted to the high levels of nutrients and turbidity that result from agriculture, especially in the lower reaches of rivers. Gaigher *et al.* (1980) noted that the lower reaches of the Berg and Breede systems were rapidly becoming devoid of indigenous fish, a phenomenon that was accredited to a combination of the fish being unable to cope with the lowered water quality, while at the same time being out-competed by alien fish that were better adapted to eutrophic river conditions. Impson & Hamman (2000) noted that often the impact of invasive alien fish has compounded the impacts of habitat degradation, driving many indigenous fish populations in mainstreams to extinction.

Finally, a serious potential threat to indigenous fish in the CFR is that of dams and weirs, built for water storage and extraction. These structures, especially large dams, are thought to have drastically affected the migration patterns of large cyprinid species (Gaigher *et al.*, 1980). The result is likely to the genetic isolation of populations that can no longer reach their natural spawning grounds within a river system (Gaigher *et al.*, 1980). Reservoirs present an added danger in that they have often been stocked with alien fish, which have the potential to invade adjacent rivers (Lintermans, 2004).

1.3.4: Invasive alien plants in the riparian zone

The question of what effect alien plants in the riparian zone may have on fish communities has seldom been broached. This is understandable, as most studies of alien trees have focussed on the damage these species are known to inflict on the terrestrial plant community. In South Africa, plants from the genera *Acacia*, *Eucalyptus* and *Sesbania* have formed dense infestations in river catchments, often causing profound changes in indigenous plant communities and soil chemistry (Ractliffe *et al.*, 2003). The spill-over effects of these alien infestations that in turn threaten instream aquatic biotas include channel-bed alterations, increased sedimentation, altered water chemistry, reduction in flow and changes in the amount and timing of allochthanous material entering the river (Ractliffe *et al.*, 2003).

All of these processes may have an indirect effect on fish communities. Changes in geomorphology may cause specific habitat such as spawning beds to disappear or become unusable to fish. Increased sedimentation results in loss of habitat for benthic fish species that need cobble-boulder substrata for cover (Bills, 1999; White & Harvey, 2001). Alien invasive riparian vegetation may however also have positive effects on indigenous fish. An increased density of woody riparian trees could in turn create more instream woody debris, which is an important source of cover for many fish species (Allouche, 2002).

Increased allochthonous loads from leaf-falls could result in larger populations of browsing and shredding aquatic invertebrates (Manicom, 1999), which in turn could provide more food for fish. Conversely, the replacement of edible indigenous leaf matter by inedible alien leaf matter could negatively impact invertebrates.

Clearly, if one were to detect any effects of invasive riparian trees on fish, one would first have to have a very good understanding of the ecological processes that drive the population dynamics of those fish.

1.3.5: Origin of this dissertation: CAPE and the Table Mountain Fund alien fish project

The conservation of the indigenous fishes of the CFR has been a pressing concern to conservators in the Western Cape area for decades, but not until recently has funding emerged to allow a proactive approach to be taken. A significant donation by the Global Environmental Facility (GEF) to the South African government in 1997 provided the key for real action to be taken. US$1 million were given as seed funding to initiate a strategic plan for conserving the Cape Floral Kingdom as a whole, including its associated marine, coastal and freshwater ecosystems (WWF-South Africa, 2000). Called the *Cape Action Plan for the Environment* (CAPE) this initiative put in place a five year strategy to better conserve the region. The plan included assessment of ecosystem health; the involvement of local communities to encourage 'buy-in' to the conservation strategies devised by CAPE to effectively conserve the region, and co-ordination between scientists, managers and stake-holders to ensure the successful implementation of projects devised by the Action Plan (WWF-South Africa, 2000). Another portion of the GEF donation was used to create the Table Mountain Fund, a subsidiary of the World Wide Fund for Nature – South Africa (WWF-SA), which would provide the financial backing for the initiation of projects that CAPE would identify as priorities, whether they be research, capacity-building, or actual implementation of management plans devised by CAPE to protect biodiversity in the CFR (WWF-South Africa, 2000).

Freshwater fish were recognised early on as an essential component of the biodiversity of the CFR, and several studies were commissioned on behalf of CAPE to assess the current biological status of all the CFR species, including their genetics, current distributions and threatened status. Reports (Impson *et al.*, 1999; Impson *et al.*, 2000) of provincial conservation agency CapeNature discuss the current conservation status of freshwater fish in the region, as well as highlighting research and conservation needs. Other reports, funded by the Table Mountain Fund (TMF), focussed on issues such as the genetic status and distributions of key species in the CFR (Bills, 1999; Bloomer, 2003). Bills (1999) examined the status of two catfish species (*A. gilli, Austroglanis barnardi* Skelton), endemic to the Olifants-Doring river system, and discovered a complex suite of genetically distinct populations, a phenomenon that appeared to indicate that these two species are very closely related, and may still be in the process of speciation. One river that both Bills (1999) and Impson *et al.* (2000) highlighted in their recommendations for conservation was the Rondegat River, a tributary of the Olifants' that contained a genetically unique population of *A. gilli* (R. Bills, E. Swartz & M. Cunningham, SAIAB, unpublished data). The lowest quarter of the river (which was separated from the upper three quarters by a waterfall barrier) was invaded by *M. dolomieu*, and field surveys indicated a loss of indigenous fish in these invaded sections (Bills, 1999). It was suggested in both reports that the river be rehabilitated by eradicating bass from the lower river and setting up a barrier weir at the lower end of the river to prevent re-invasion. The river was also seen as an opportunity to quantify for the first time in detail the effects of smallmouth bass on indigenous fish species within the CFR. As a result, two MSc projects were funded by WWF-SA to study the effects of *M. dolomieu* on fish and on invertebrates in the Rondegat River. This dissertation describes the results of the first of these two projects. The other project deals with the potential impacts of *M. dolomieu* on the invertebrate community of the Rondegat River.

The problems that generally face scientists attempting to quantify alien species impacts in rivers are threefold: (a) baseline data on the affected indigenous species are sparse if not non-existent; (b) the alien species is generally not the only potential impact present in the system, with rivers having suffered from habitat alterations both before and after the introductions; and (c) introductions are generally not followed by an ongoing monitoring program to determine impacts on the local community after the introduction (Crivelli, 1995). It was therefore important, for this project to be effective, that these factors be taken into account. In the case of baseline ecological data, some unpublished information was available from Dr Jim Cambray of the Albany Museum on physical habitat preferences of the indigenous species. There were also some data available on the feeding ecology of the Clanwilliam yellowfish (Van Rensburg, 1966). However, in terms of the other indigenous fish species, this project would significantly expand our limited knowledge of their feeding ecology. The objectives of this project are detailed in Section 1.4 below.

1.4: THE RONDEGAT RIVER: AN OVERVIEW

1.4.1: Location and Geography

The Rondegat River is a tributary of the Olifants River, and is situated in the Cedarberg mountain range, part of the Cape Fold Mountains of the Western Cape. It has a fairly small catchment of approximately 111km^2 in area (February, 2002) and has its origins in three tributaries rising in the Ou-Uitkyk pass and joining to form the Rondegat River approximately 4km upstream of a campsite called Algeria (Figure 1). It then runs north-west for 20km before veering south-west into a narrow canyon, after which it flows into the Clanwilliam Dam reservoir on the mainstem of the Olifants River. During the period of fieldwork, the water level of the reservoir fluctuated dramatically, so that for half the time the Rondegat had a true confluence with the Olifants, while for the other half it flowed directly into the reservoir. The river descends 1000m from its source in the Ou-Uitkyk pass to its confluence with the reservoir and appears to flow through three river zones (*sensu* Brown *et al.*, 1996) (Figure 2). From its source to approximately 4km downstream, the river falls within the mountain stream zone. The river is very narrow (approx. 2m wide) and steep here, with a mean gradient of 0.27, and its banks are densely populated by afromontane and fynbos riparian trees. The river then enters a foothill zone, characterised by a mean gradient of 0.02 (Brown *et al.*, 1996), which it maintains more-or-less to the point where it meets the high-water mark of the reservoir.

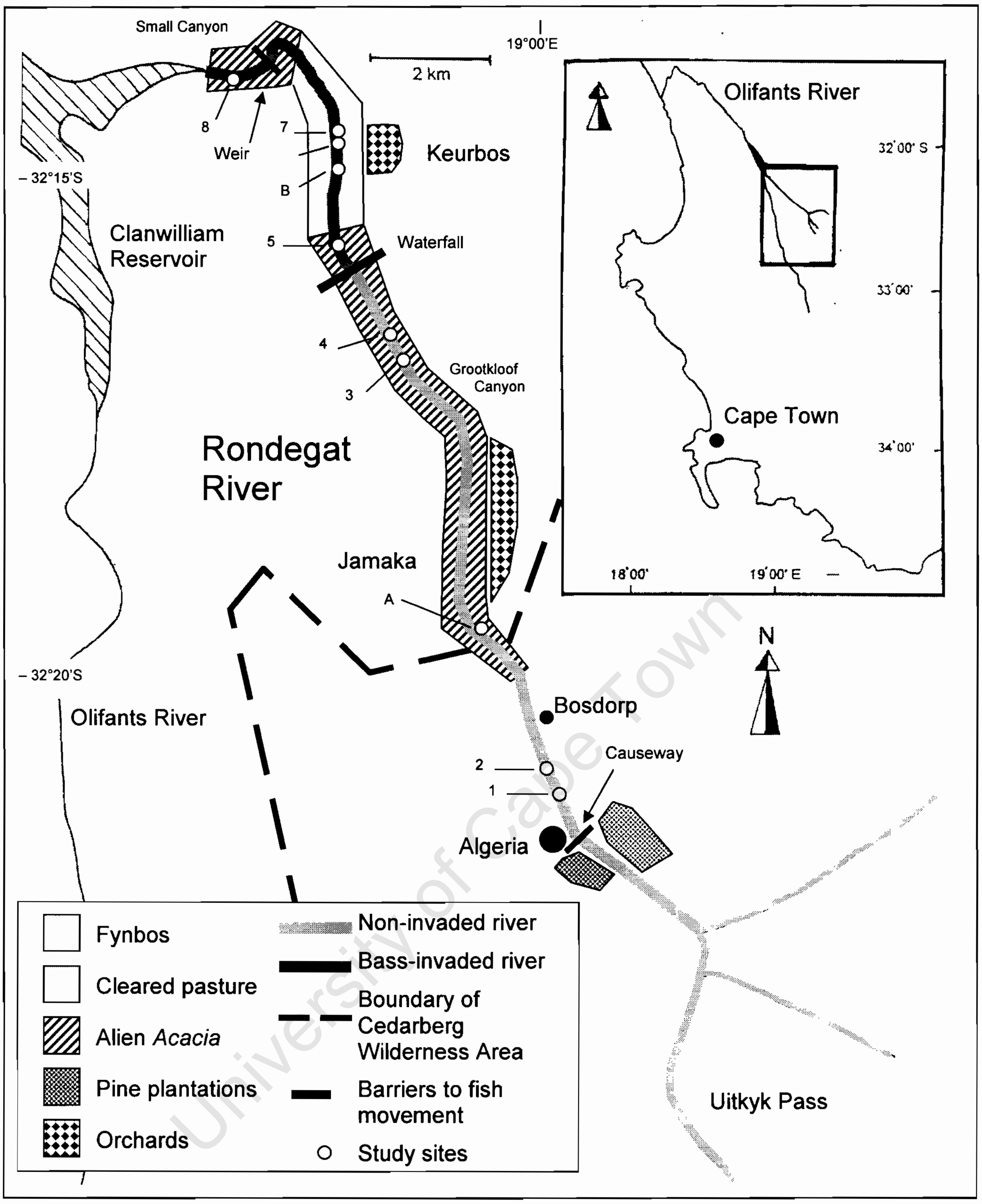

Figure 1: Map of the Rondegat River showing study sites, major land-use patterns in the catchment and critical barriers to fish movement.

The river within the foothill zone is characterised by a single alluvial channel with a boulder-cobble dominated substratum. The channel has been modified in places by the local farmers, who have dug furrows and diversion channels and built weirs in several places to feed flood irrigation systems and increase the accessibility of water to their cattle, which freely roam the floodplain and have created many trails that cross the river channel. The middle reaches of the river run through a large canyon (the *Grootkloof*, after which the local farms are named) and at the end of this canyon, the river plunges down a short, steep bedrock cascade, over a short waterfall and into a deep pool. This waterfall is the largest natural barrier to fish movement on the river (Appendix 3; Plate 12), and is approximately 5km from the high water mark of the reservoir, which is also where a road-bridge crosses the river (Figure 2).

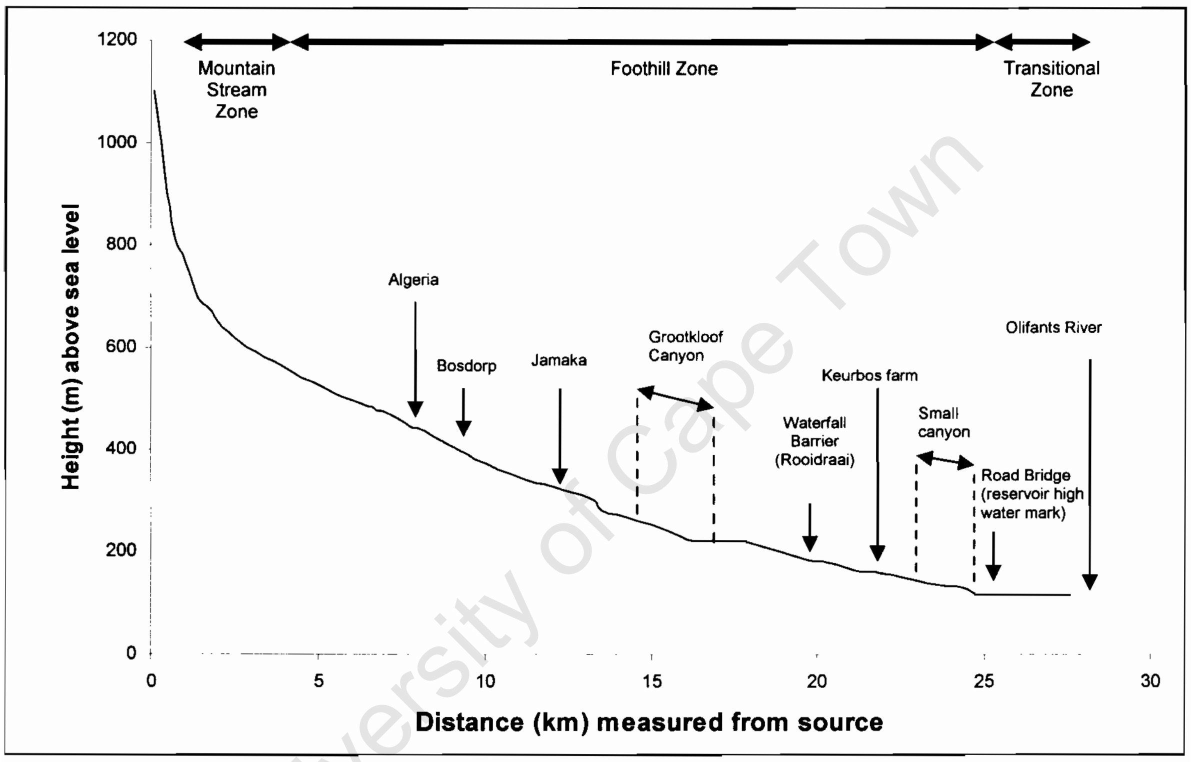

Figure 2: Longitudinal profile of the Rondegat River showing the three putative geomorphic zones and key features on the river.

From the waterfall, the valley flattens out again and the river forms a floodplain, approximately 150m wide, which continues until the river turns southwest and enters the

small canyon. The river forms a straight narrow channel that hugs the south-western edge of the floodplain, while the rest of the floodplain has been converted to cattle pasture.
In the canyon, although there is no perceptible change in gradient, the river shows signs of rejuvenation, with riffles becoming scarcer and bedrock chutes and cascades more common. When the river leaves the narrow canyon at the high water mark of the reservoir, it enters the floodplain of the Olifants River. Here the gradient drops off considerably to around 0.003 and, when the river is not inundated, it shows the typical mixed sand-cobble-bedrock substrata of a transitional zone river (Brown *et al.*, 1996), before joining the Olifants mainstem. For the purposes of this study and in order to limit the effect of longitudinal variation when comparing sites, it was decided that all fieldwork would be conducted within the foothill zone, from Algeria to above the high water mark of the reservoir.

1.4.2: Land use and alien vegetation in the Rondegat catchment area

The first 10km of the Rondegat River flow through the Cedarberg Wilderness Area (CWA) (Figure 1). This wilderness area is managed by CapeNature, and is part of the Greater Cedarberg Biodiversity Corridor, one of three proposed 'Mega-reserves' in the CFR. It is covered for the most part with near-pristine mountain fynbos. Environmental impacts in these reaches of the Rondegat River include guest cottages in the mountain stream zone at Uitkyk, a campsite and office buildings at Algeria, the management centre of the wilderness area, as well as the Bosdorp, a settlement of 40 housholds (February, 2002). The Bosdorp houses nature conservation field rangers and part-time employees of government programs, as well as their families. A feature of the river as it flows through the campsite at Algeria is a causeway that also serves as a weir, forming a swimming pool for campers. This causeway has a sluice that is opened in the winter in order for the river to flow, but in summer it forms a major discontinuity in the river. Historically, substantial pine (*Pinus radiata*) and eucalyptus (*Eucalyptus* sp.) plantations were present in the upper reaches of the catchment. The vast majority of these plantations had been removed from the catchment by nature conservation authorities by the time sampling commenced. However, bare ground left behind by the removal of pine plantations, coupled with recent fynbos fires in the upper catchment, meant that erosion of topsoil in the upper catchment had probably been higher than average in recent times (February, 2002). Heightened erosion had the potential to increase sedimentation in the Rondegat River.

Downstream of the wilderness area, the Rondegat River flows through four farms. The upper three farms are subdivisions of a single farm, and so all bear the name Grootkloof. Farming activities in the catchment focus on citrus and cattle (February, 2002). A campsite by the river on the uppermost farm, called Jamaka (Figure 2), has a temporary weir that creates a swimming pool for campers in the summer. From when it leaves the wilderness area, until after it enters the furthest downstream farm, Keurbos, 20km from the source, the river's riparian zone is invaded by the Australian black wattle (*Acacia mearnsii*) and, to a lesser extent, blackwood (*Acacia melanoxylon*). Other alien plants such as bramble (*Rubus* sp.), gum (*Eucalyptus* spp.), Port Jackson wattle (*Acacia saligna*) and hakea (*Hakea* spp.) are also present in small quantities in the riparian zone (February, 2002). The farmer at Keurbos, Mr. J.H. Nieuwoudt, has cleared large sections of the alien infested riparian zone on his property, in order to create more grazing land for his cattle and to harvest the wood for sale (February, 2002). As a result, the section of river from the waterfall barrier (which is near to the boundary between Grookloof and Keurbos farms and the site of a holiday cottage called Rooidraai) downstream to the canyon is characterized by a pasture-dominated floodplain and an extremely narrow riparian strip, consisting of a mixture of indigenous plants and alien recruits.

1.4.3: The fish community of the Rondegat River

The Rondegat River has particular significance for fish conservation in that it is a tributary of the Olifants-Doring system, which is the hotspot for fish diversity and endemism in the Western Cape and nationally (Skelton *et al.*, 1995). In all, the catchment contains eight fish species that are endemic to it, all of which are threatened. It is therefore an appropriate location at which to study the factors that threaten fish conservation in the Western Cape region.

The Rondegat's fish fauna includes: two endangered species, the Clanwillian redfin (*B. calidus*) and the fiery redfin (*P. phlegethon*), and two vulnerable species, the Clanwilliam rock catfish (*A. gilli*) and the Clanwilliam yellowfish (*L. capensis*), as well as the near-threatened Cape galaxias (*G. zebratus*), which is present in the upper reaches. The status of the galaxiids is currently under review, as it is now believed that the populations of this species found in the Olifants-Doring system belong to three genetically distinct species, which will have a more threatened status once they are recognised (Swartz, 2003). Since the "Cedarberg galaxias" has yet to be officially described, it will continue to be referred to in this dissertation as *G. zebratus*. Clanwilliam sawfin (*Barbus serra* Peters) were once found in the lower reaches of the Rondegat (Van Rensburg, 1964) but are no longer present there (Bills, 1999). Smallmouth bass (*M. dolomieu*) invaded the lower Rondegat River after having been stocked in the Olifants River at Keerom in 1945 and have probably been resident since the 1950s, when they invaded Clanwilliam dam (Harrison, 1963). They have penetrated upstream as far as the waterfall at Rooidraai, approximately 7km upstream of the confluence with the Olifants River (Figures 1 & 2). Only this natural barrier appears to prevent the bass from invading further upstream.

1.5: OBJECTIVES OF THIS DISSERTATION

The impact that alien fish have on indigenous fish species is often easy to infer from circumstantial evidence, but generally harder to demonstrate empirically, due to the myriad of environmental factors affecting the fish. The key to identifying the impact of bass therefore is to find which other factors play a significant role in the health and distribution of the indigenous fish, and how these may directly or indirectly affect any interactions between the fish species. It was decided that the most effective way to approach this problem would be to investigate alternative factors thought *a priori* to influence fish distributions and abundance.

Physical habitat quality, and in particular the degradation of instream habitat, which is known to seriously affect some fish species, would be studied in as much detail as was logistically viable in order to assess its effect on current fish distributions. Food availability, which can influence both the distribution of a fish species, as well as its ecological interaction with other species through competition, was also acknowleged in the planning stages of the project as a factor to be assessed. It was, however, decided that due to the amount of data already being collected, that this aspect of the river's ecology would be looked at in less detail than bass and habitat impacts. In particular, the assessment of food source abundance, including invertebrate, plant and other organic material on more than a semi-quantitative basis, was seen as being beyond the scope of the current study, though every attempt was made to ensure that meaningful interpretation of the data gathered could be made.

By assessing the influence that habitat, and to a lesser extent, food availability had on fish distributions, one could hope to uncover and filter out the "ecological noise" that might obscure the true extent and nature of bass impacts, while at the same time learning which factors might have exacerbated or mitigated against the impacts of bass. It could also allow one to investigate the relative impacts of alien invasive riparian trees on indigenous fish, as these species would be expected to affect fish indirectly through the alteration of both habitat and food availability.

The first aim of fieldwork was to gain a working understanding of the population status and distributions of indigenous fishes within the Rondegat River. By quantitatively surveying fish populations at several sites on the river during different seasons, the severity of bass impacts could be assessed.

In order properly to quantify these impacts relative to alternative environmental impacts, the following null hypotheses were tested:

H_0A: Measured aspects of physical habitat do not differ between bass-invaded and non-invaded sites.

H_0B: If some aspects of physical habitat do differ between bass-invaded and non-invaded sites, then none of the indigenous fish species are limited in their distributions by these factors.

H_0C: No species of indigenous fish show a preference for a single limiting food type.
H_0D: If some species of indigenous fish do prefer certain food types, the availability of these food types does not differ between bass-invaded sites and non-invaded sites.

If data analysis could not reject one of H_0A or H_0B, and likewise one of H_0C or H_0D, then the presence of bass could be considered to be the overriding factor affecting indigenous fish populations at bass-invaded sites.

In order to test H_0D, gut-content data would be gathered for all four endangered fish species. An additional spin-off of this analysis would be to get a clearer understanding of the ecological interactions each species has with its environment, and how vulnerable each may be to indirect effects of habitat quality through local food webs. *Micropterus dolomieu* would also be captured for stomach analysis, firstly to look for evidence of direct predation on indigenous fishes, but secondly to compare other taxa found in the stomachs with those found to be eaten by indigenous fish species, so that the potential for food competition could be assessed.

The final objective of this dissertation was to use the findings of the study to compile a comprehensive management plan for the Rondegat River, in order to aid the successful implementation of the rehabilitation recommendations originally proposed by Bills (1999) and Impson *et al*. (2000).

CHAPTER 2: METHODS

2.1: THE CHOICE OF STUDY SITES – SEEKING APPROPRIATE REPRESENTATION AND COMPARABILITY OF HABITAT

When choosing sites at which to perform fish surveys, several issues were taken into account. Firstly, it was important to design the site layout to ensure representation of the variety of physical habitat present within the catchment. Biological patterns, such as the distributions or diversity of fish communities, are linked to physical patterns of the river they inhabit, and when trying to understand these interactions it is crucial to examine the role of both time and scale (Frissel *et al.*, 1986).

In terms of time, it was decided to sample over the period of a single seasonal cycle, beginning in the early austral spring of 2003 and ending in the autumn of 2004. The month of September was originally chosen as the early spring month in which to begin sampling. However, exceptionally cold and wet conditions in August 2003 meant that in September the river was experiencing the equivalent of mid-winter high flows, with snow-melt from the mountains keeping temperatures between 10 and 15°C (Appendix 1). Fish surveys were almost impossible to perform, and an incomplete and unreliable data set was gathered. As a result, data gathered from September has for the most part been excluded from the quantitative and statistical analyses presented in this thesis, although what was gathered during the survey is still referred to in Chapters 3 and 4. Successful surveys were conducted in October 2003, November 2003 and April 2004. October was effectively mid-spring, while late November represented summer, the season in which indigenous cyprinid spawning is known to occur (Skelton, 1987). April represented late autumn, just before winter low temperatures began to alter fish behaviour (and consequently their visibility) and when the young-of-the-year or age 0+ (*sensu* Garner, 1996) cyprinids would be big enough to be visually identifiable.

Two fundamental concepts to understand when choosing a sampling regime for surveys are those of habitat scale and grain. Scale is the size of area surveyed (Vadas, 1992) and the choice of scale varies according to the nature of the organism being studied. Grain is the sampling resolution used within the area (Vadas, 1992), and is important when trying to isolate particular interactions between fish and their environments. For example, to get a broad idea of a species' relationship with flow, one could measure the overall current velocity of each study area, by timing fluorescent dye as it moved downstream through the study site (Townsend & Crowl, 1991). If one wanted to capture the specific flow preferences of a species, one would need to measure point velocities where each fish was captured or observed (King & Tharme, 1993).

While addressing the questions of scale and grain, logistics played a substantial role in dictating the limits of study-site size and number. *Austroglanis gilli* is a benthic species (Skelton, 2001), and the only non-destructive way to sample this species effectively by day is by electrofishing (Bills, 1999). Quantitative electrofishing is a complex process and its use in this project is discussed in more detail in section 2.2.1. However, an implication of using electrofishing, was that a reasonable area of river in which to sample had to be decided upon. Many fish surveys that have employed electrofishing have sampled large river-reaches, spanning up to 100 meters (Kruse *et al.*, 1998; Pires *et al.*, 1999). The goal of such surveys is to sample all available geomorphologic habitat units present in a given reach of river.

Wadeson (1994) highlighted the fact that considerable confusion exists in how to properly define these geomorphologic units. The units most commonly referred to in studies involving the study of fish-habitat interactions are riffles and pools (Gorman & Karr, 1978; Rodrigues, 1995; Martin-Smith, 1998) although many studies recognise more (Vadas, 1992; Baran *et al.*, 1997). The universal use of riffles and pools in the literature is probably a result of the fact that they are easy to define visually: riffles are generally defined as shallow reaches with rapid flow, while pools are defined as slow, deep reaches, although the exact definition varies widely in the literature (Wadeson, 1994). While many studies refer to these physical features simply as "habitat units" (Martin-Smith, 1998, Mullner et al., 1998) they are also known as "mesohabitats" (Vadas, 1992; Baran *et al.*, 1997), in that they fall between the reach, or macrohabitat scale, and the smaller microhabitat scale, which is defined as a section of a pool or riffle with a homogenous depth, flow and substratum (Frissel *et al.*, 1986).

The problem with sampling an area that includes both riffles and pools is that electrofishing can prove difficult in pools where the water is too deep for hand-held electrofisher units to operate effectively (Joyce & Hubert, 2003). Since in this study I had access to only a hand-held unit, it was decided that two different techniques would be needed to sample riffles and pools, which were the two most common mesohabitats present in the river. Study sites would therefore be split into paired riffle and pool sites, with electrofishing performed in riffles, and snorkelling surveys performed in pools. Many studies conducted on trout have found the two techniques to be comparable in terms of the numbers of fish they detect (Mullner *et al.*, 1998; Wildman & Neumann, 2003). This is because, given appropriate attention to technique and favourable habitat conditions (e.g. low densities of woody debris at a site), they have provided highly concordant fish survey data when used to survey the same reach of river (Wildman & Neumann, 2003). This suggested that the two techniques could be used to complement each other when only one or the other technique could be used.

In selecting paired pool/riffle sites, several issues needed to be considered. From a logistical point of view, the site had to be accessible by vehicle, because of the large amount of heavy equipment that was needed, especially for electrofishing surveys. Consequently, sites had to be within short walking distance of a road, or else accessible by a 4-wheel-drive vehicle. Once these constraints were accommodated, the most important criterion was to select replicates of the various biological invasion scenarios found down the length of the river. These invasion scenarios could be described as:

A) Near-pristine fynbos riparian zone; no alien fish
B) Alien-tree-invaded riparian zone; no alien fish
C) Alien-tree-invaded riparian zone; alien fish present
D) Riparian zone converted to cattle pasture; alien fish present

A pilot study was conducted in May 2003 to select potential study sites, identify access points to the river and get an initial impression of fish distributions within the river. Twelve sites were selected, but during the first full field trip in September 2003, this number was reduced to eight, a number that could be surveyed within the 9 survey days that funding enabled me to be in the field. Two additional sites referred to as "A" and "B" were added in later field trips to provide additional information into fish distributions and habitat uses. Locations of study sites are illustrated in Figure 1, while their features are displayed in Table 1.

Table 1: List of study sites visited during fieldwork. Notes indicate unique positional characteristics of sites and features that deviated from initial definitions for pool and riffle habitats. Photographs of sites are shown in Appendix 3.

Number	Name	GPS	Invasion Scenario	Notes
1	Algeria Bridge	32° 22' 12" S 19° 03' 20" E	A	
2	Rockface	32° 22' 06" S 19° 03' 12" E	A	Pool very small (16m* 8m)
3	Meadow	32° 17' 35" S 18° 59' 51" E	B	Pool very shallow
4	Cottage	32° 17' 20" S 18° 59' 39" E	B	
5	Rooidraai	32° 16' 40" S 18° 58' 46" E	C	Approx. 100m downstream of barrier falls. No riffle locally available Electrofishing site a shallow bedrock run
6	Upper Keurbos	32° 15' 53" S 18° 58' 18" E	D	Electrofishing site a combination of riffle and deep sandy run
7	Lower Keurbos	32° 15' 48" S 18° 58' 15" E	D	Riffle immediately downstream of Upper Keurbos Pool
8	Canyon	32° 15' 30" S 18° 57' 12" E	C	Separated from other sites by abstraction weir. riffle and pool approx. 500m apart
A	Jamaka Pool	32° 20' 32" S 19° 01' 25" E	B	Artificial pool, no riffle. Visited only in April 2004
B	Extra Keurbos Riffle	32° 15' 58" S 18° 58' 21" E	D	Riffle, no pool. Visited in November 2003 and April 2004

2.2: FISH SURVEYS

2.2.1: Snorkel sampling

Visual underwater sampling of fish using a mask and snorkel is a very attractive method for assessing fish stocks. Unlike removal methods such as electrofishing or netting, snorkelling does not require much equipment, take much time to perform, or have the same potential to harm fish (Mullner *et al.*, 1998). It also has added benefits, in that microhabitat utilisation and behaviour of fish can be observed (Surface Water Resources Inc., 2003). Snorkel sampling does however have unique drawbacks that must be considered. Increased turbidity can make observations difficult; increased flows can cause safety hazards; fish weights and accurate measures of fish length cannot be obtained; and the use of more than one diver can increase the possibility of double counting (Surface Water Resources Inc., 2003). In this project, snorkelling was chosen early on as the most appropriate method for sampling fish in pools, as it would allow for relatively quick sampling with minimal equipment, and was likely to be reasonably accurate due to the generally good visibility found in the river, with the bottom visible at all times (pers. obs. 2003). In order to improve the accuracy of fish counts, three pass snorkelling (surveying the study area three consecutive times) by a single diver was used at all pool sites.

Northcote & Wilkie (1963) found that repeated snorkel counts of salmonids and cyprinids in the Similkaneen River, Canada, gave reasonably similar population estimates. This result suggests that a second and third pass should not overly bias the accuracy of the count by severely altering the behaviour of the fish. By calculating the mean number of each species counted in the three passes, one could obtain a more accurate estimate of the true population present in the pool than one generated from a single pass. While more than three passes would have improved the statistical rigour of the count, time constraints in the field prevented this. A drawback of this approach is that there would be no way to assess the accuracy of the mean statistically, as this would entail repeated 3-pass sampling of the site to build up replicate means with which to compare the original mean (A. Govender, UCT, pers. com. 2004). As this was logistically impossible to achieve during field work, the efficacy of snorkelling was not tested in the field.

During fieldwork, all pools were surveyed by a single diver. The diver entered the pool at its downstream end and proceeded to swim slowly upstream in a zig-zagging pattern to maximize coverage of the pool's area (after Mullner *et al.*, 1998). Special care was taken in examining the margins of the pool, as well as any woody debris, which often provided cover for the fish. As soon as fish were seen, the diver would approach to within approximately 1 m of them before recording species and size class on a white Perspex slate with pencil. Each size class was a categorical measure of total length in cm, which was estimated using a yellow 30cm ruler attached to the writing tablet, held at arm's length to adjust for underwater magnification. Each species was allocated at least four size classes, which were calibrated to define key life history stages such as larvae, juveniles and adults (Table 2). The size limits of the life-history stages of each species described by King and Tharme (1993) were used in choosing size classes. With cyprinid larvae, underwater identification was generally impossible, and these fish were simply noted as larvae.

On conclusion of the three passes, visibility was measured using a Perspex board with a letter "A" written in black permanent marker (after Mullner *et al*, 1998). Unlike the method described by Mullner *et al* (1998), the decision was made to measure visibility at the end of the survey rather than the beginning, so that the fish in the pool, which was sometimes a very confined habitat, would not be unnecessarily disturbed prior to sampling.

Table 2: Size classes (cm TL) and corresponding life-history categories used to estimate fish sizes during pool surveys (adapted from King & Tharme, 1993)

Species	Larva	Small juvenile	Juvenile	Large juvenile	Small Adult	Adult	Large Adult
*Labeobarbus capensis**	0 – 2	2 – 5	5 – 10	10 – 20	20 – 30	30 – 40	> 40
Barbus calidus	0 – 2	–	2 – 5	–	5 – 8	8 – 11	> 11
Pseudobarbus phlegethon	0 – 2	–	2 – 4	–	4 – 7	7 – 10	>10
Austroglanis gilli	–	–	< 8	–	–	≥ 8	–
Micropterus dolomieu	–	< 5	5 – 10	10 – 15	15 – 20	20 – 30	> 30

*While King & Tharme (1993) list 30cm as the minimum size for mature *L. capensis*, the size frequencies encountered in the yellowfish population during the Autumn pilot survey led me to believe that Rondegat yellowfish could be mature by the time they reached the 20-30cm size class. The fact that stunted adults have been encountered in the Clanwilliam sawfin, another large Olifants River cyprinid (King & Tharme, 1993), supports this assumption.

2.2.2: Electrofishing

Electrofishing surveys that seek to gain quantitative abundance data usually involve mark-recapture or multiple-pass removal electrofishing to calculate an estimate of true abundance at a study site. While both these techniques are well regarded and widely used, they both involve a considerable time spent sampling, as a site essentially needs to be sampled twice, and more often four times (Kruse *et al.*, 1998). In planning field techniques for this project, the original intention had been to use three-pass removal sampling (after Cowx, 1983) over a set length of riffle reach nearby to the pool at each site. During the September 2003 field trip it however became clear that in order to sample an average of three habitats a day (a work regime imposed by the project's time and budgetary constraints) it would be necessary to restrict electrofishing surveys to a single pass. Bills (1999) found that while single-pass surveys were seldom sufficient in detecting total abundance of *A. gilli*, efficacy was greatly improved by electrofishing relatively short reaches of river, working downstream into a seine net. This technique was adopted as the basis for this project's methodology.

Due to the large variability in riffle length throughout the river, a standardised 20m sub-sample of riffle area was sampled. Each 20m reach was sub-divided into four sub-sets by moving a seine net between stationary block nets, sequentially sampling different fractions of the area between the block nets. A mean fish density per sampling block could then be calculated for the 20 meters by averaging the number of fish caught by the shocker operator and in the mobile seine net during each shocking pass. It was assumed that this method would maximise the efficiency and reliability of fish density estimations, within a given comparable subset of the habitat. Escapement (i.e. fish not caught within the blocks or in the mobile seine net) would be measured by placing a fyke net immediately downstream of the site, into which fish flushing downstream would swim and be trapped. By measuring the escapement of each species, a rough idea of an individual fish's probability of capture could be gained, providing that the fish in the sampling area were randomly distributed at commencement of shocking.

In order to minimise downstream escapement from the 20m sampling reach during set-up, the fyke net was set downstream first, followed by a fixed seine net at the top of the site. Both the seine and fyke nets were secured to the riverbed by placing boulders on the netting and ensuring there were no obvious gaps between the net and substrata. This ensured that the site was reasonably contained and that fish could not escape except into the fyke net. The site was then subdivided into four sampling blocks, the divisions being in a relatively deep part of the riffle where a seine net and its bag could be laid open in the current. A 5m x 2m seine was then laid across the boundary line between blocks 1 and 2 (block 1 being the furthest upstream). Once all nets were in place, the shocker operator, using a DEKA 3000 backpack electrofisher, began to shock the riffle, moving downstream in a zig-zagging motion towards the mobile seine. The shocker was set to 600 volts so that, depending on the conductivity of the site, it could discharge a current of between 4 and 6 amps into the water between the two electrodes. The waters of the Rondegat River have very low conductivity (Appendix 1), and as a result the current dissipated rapidly in the water around the electrodes, so fish were generally not stunned until they passed close to an electrode. The method was designed so that the fish that were not caught by the probe would be flushed into the mobile seine.

Any fish captured by the probe-net were immediately placed in a bucket with water. Once the first block had been shocked, the portable seine was lifted, and all fish that had floated or swum into the net were placed in the bucket. This procedure was repeated for the other 3 blocks, a new bucket being used each time. Finally, the fyke net was removed and fish that had swum into it during sampling were placed in a fifth bucket, representing "extra-catch". The fish in each bucket were separately weighed on a battery powered digital scale to the nearest gram, and measured in cm to total length (TL) on a fish measuring board. Fish were held in the bucket until they were judged to have fully recovered from electronarcosis before weighing and measuring, and subsequently were immediately released back into the riffle.

Since this method employed a single electrofishing pass to estimate local fish abundance, the question arose during sampling as to the efficacy of the method. While multiple-pass electrofishing is the standard method recommended for fish survey work, the use of single-pass electrofishing as a practical alternative where time constraints are inhibitive has been explored by researchers (Kruse *et al.*, 1998; Meador *et al.*, 2003). In order to assess the efficacy of this project's single-pass method, the sampling procedure was performed twice on the final field survey in April 2004, so that a second-pass data set could be generated. Fish that were captured in the first pass were released downstream of the sampling area to prevent re-capture. The efficacy of the first pass to capture fish was assessed using a probability of detection equation (after Meador *et al.*, 2003):

$$Pj = (n_1j - n_2j) / n_1j$$

Where: Pj = Probability of capture for species j
n_1j = Number of species j captured in pass 1
n_2j = Number of species j captured in pass 2

In order to assess the effectiveness of single-pass electrofishing for each species, their mean Pj was calculated from all sites where the species was captured. To assess the confidence one could place in these means, the number of times the removal model failed to generate a result (i.e. when there were greater or equal numbers of a species caught on the second pass) was also examined. The results of this analysis (Table 3) show that *A. gilli* displays a relatively poor Pj of 53.3%, indicating that one could expect on average to catch half of the catfish present at a site on the first pass. While this could be considered a poor outcome, the failure rate of 0% indicated that this was relatively consistent capture efficiency, and could be accepted with high confidence. On the other hand, *B. calidus* had a high probability of capture (87%), but a model failure rate of 75%, indicating that electrofishing failed to capture the majority of the local population on the first pass most of the time, and therefore the field data generated for this species could not be accepted as representative with high confidence. Meador *et al.* (2003) used this method to estimate the efficacy of single-pass electrofishing to assess community structure in American mountain streams, and found the failure rate in detecting all species present in the first pass to be 40%, which they considered high.

Table 3: Mean probabilities of capture and removal model failure rates for 2-pass removal sampling conducted at riffles sites during the April 2004 fieldtrip.

Species	Mean probability of capture (%)	Removal model failure rate (%)
Austroglanis gilli	53.3	0
Barbus calidus	97.7†	75
Pseudobarbus phlegethon	75.5	25
Labeobarbus capensis	60†	75
Micropterus dolomieu	100*	0*

† Based on only 1 valid result of the removal model

* Based on only 1 survey site record

In the end, this test did not have enough replicates (maximum 4) to provide a definitive answer to the question of single-pass efficiency, but it does show that the single-pass data generated for *A. gilli* and *P. phlegethon* in previous field trips were inaccurate at best, while in the case of *L. capensis* and *B. calidus*, the survey data were probably only useable to indicate the species' presence at a site. The efficacy of electrofishing in counting bass could not be verified here, since they were only caught at one riffle site in April (Table 3). It may have been better in hindsight to utilise 3-pass depletion to assess the efficacy of electrofishing, as this would have provided a variance for the probability of capture (I.G. Cowx, University of Hull, pers. com. 2005), but the lack of time available in the field ultimately prevented this approach from being taken.

While it is important to identify potential flaws in the data set, the problem facing this study, as with any study dealing with threatened species in restricted ranges, is that it is seldom possible to sample widely and intensely enough to generate data with high confidence levels. The Rondegat River is one of very few rivers in the region where research on indigenous fish can be carried out easily, and the other rivers do not support the same diversity of indigenous fish species. While logistics was the major motivation behind limiting this study's electrofishing surveys to one pass, and the total number of riffle sites to eight, it may also have become unsound from a conservation point of view to sample these fish at the intensity and replicate number suggested by studies in the literature. For instance, *A. gilli* stunned by electrofishing but not captured often fall prey to crabs (R. Bills, SAIAB, pers. com. 2004), and so including multiple passes and added sites may have lead to increased mortality.

The implication of these findings was that fish abundance data generated by pool and riffle surveys could not be compared when linking them to habitat variables generated for those sites, as there was no way of comparing the two methods' relative accuracy. However, since the sampling method was consistently applied both in pools and riffles, the relative abundance of fish among pools can be considered comparable on the basis of equal sampling effort. As a result, the relative mean abundance of each species, except for the benthic *A. gilli*, was analysed using the pool data. The same reasoning could be used to justify the comparison of riffle data, at least for the species consistently detected by electrofishing. Only riffle survey data for *A. gilli*, *P. phlegethon* and *M. dolomieu* were utilised, and due to the low numbers of fish captured, the total number of fish captured at each site was used in analysis rather than a calculated density-per-sampling-block.

2.3: LINKING FISH DISTRIBUTIONS TO FOOD AVAILABILITY

2.3.1: Field assessment of food availability

In order to investigate the role food availability might play on fish distributions, several different methods were employed to obtain a working understanding of the relative abundances of food items available to fish in the Rondegat. The composition of the aquatic invertebrate community was assessed by conducting kick-sampling surveys at each site using the methods of the South African Scoring System biomonitoring protocol (Dickens & Graham, 2002). Invertebrates captured were identified to family, and these data were used for comparisons with stomach content analysis. While a large proportion of aquatic invertebrates are captured by fish feeding from the benthos of a river, many invertebrates, both aquatic and terrestrial, are captured while drifting downstream, either trapped on the surface or swimming in the water column. In order better to understand the drift component of invertebrate food, plankton nets were used at four sites during each field survey to capture drift. Each net was placed upstream of the fish-survey site and left for 24 hours to gather both day and night drift. Samples were fixed in 5% formalin and later transferred to 70% ethanol. Each drift sample was sorted in the laboratory in a two-part method. Firstly, the entire sample was visually examined for 30 minutes and all invertebrates visible with the naked eye were removed. Next, a handful of the sample was removed as a sub-sample to allow more detailed analysis. Each sub-sample was examined under a microscope and all invertebrates present were removed for identification.

Benthic detritus is a potentially important food source for both fish and invertebrates in rivers. Filamentous algae are known to be important in the diet of small cyprinids in the CFR, including the Eastern Cape redfin (*Pseudobarbus afer* Peters) (Skelton, 1993), and the Burchell's redfin (*Pseudobarbus burchelli* Smith) (Shelton, 2003). Coarse particulate organic matter (CPOM) includes leaves and other allochthanous plant material that enters the river from the riparian zone, and forms a crucial food source for certain feeding guilds in the invertebrate community (Davies & Day, 1998). Detritus, CPOM and algal abundance were all assessed visually during instream habitat assessments. These visual assessments are discussed in Section 2.4.2.

2.3.2: Diet analysis on fish species

When studying the potential effects of bass as food competitors with indigenous fish, it is imperative that the feeding ecology of both the indigenous fish and the bass be sufficiently understood. Representative samples should be taken in all seasons, so that seasonal changes in diet can be examined.

Fish were captured for stomach analysis in three ways. In October 2003 and April 2004, sub-samples of each species were taken from those caught in the electrofishing surveys. No fish were removed during the November electrofishing surveys, as it was thought that removing fish repeatedly from the survey site would negatively bias fish densities in the final survey. In November and March, fish were caught by seining a pool downstream of Site 4. In October, November, and April, fyke nets were set overnight at the pool at Site 5, as well as in the pool directly downstream of the waterfall barrier, and bass, yellowfish and some catfish were caught in this way.

All captured fish were preserved in 10% formalin. I attempted to process all fish within two hours of capture, so that digestion would not have a serious effect on the stomach contents. In the laboratory, two methods of analysis were used, depending on the species. In the case of bass and catfish, the stomach only was removed and dissected. Cyprinids, on the other hand, do not have a true stomach, but instead have an enlarged first intestinal loop, known as the pseudogaster (Eccles, 1985). Van Rensberg (1966), when studying the diet of *L. capensis*, examined only the pseudogaster, which in adult fish is easily distinguishable from the rest of the alimentary canal, and large enough to provide a substantial sample. However, since most of the yellowfish and all the redfins collected in this study were small and had partially or completely evacuated pseudogasters (often a result of the delay between capture and preservation), the contents of the entire gut were analysed.

Whether studying a stomach or an entire gut, a standard sorting protocol was established. First the gut was dissected down its entire length using scissors or forceps, and all contents were emptied into a Petri dish. Using a binocular dissecting microscope, all contents were sorted into major food item groups. These groups comprised crab, other aquatic invertebrates, fish, detritus, filamentous algae, and woody plant material. These components were each stored separately in glass vials in 70% ethanol. In order to assess the relative contributions of these categories to total food intake, volumetric analysis was then conducted on the individual food category components of each gut sample using the formula:

$$V = X.\ \Pi r^2.\ H$$

Where: V = volume of food component
Π = 3.1415926
r = radius of the glass vial base
X = estimated proportion of vial base surface area covered by food component
H = mean height of food component inside vial

For each categorised sub-sample, the percentage of the tube's base surface area (Πr^2) covered by food material was visually estimated, after which the mean vertical height of the food material (H) was measured with Vernier callipers. The mean percentage of total gut volume contributed by each category was then analysed to get a better understanding of the food preferences of the fish. The presence of sand and parasitic worms (Classes Nematoda and Cestoda) was also noted in each sample.

2.4: LINKING FISH DISTRIBUTIONS TO HABITAT VARIABLES

2.4.1: Selecting a useful method

When selecting a method to measure fish habitat, it is critical to decide what aspects of habitat are of importance to the fish. Three fundamental aspects of physical habitat in rivers that fish are believed to interact with are depth, current velocity and bottom type (Owen & Karr, 1978). All of these physical factors, which consist of interacting aspects of geomorphology and hydrology, can be combined in the general concept of cover. Cover can be defined as aspects of physical habitat that perform three major functions in the lives of fish. These are anti-predation, visual isolation, which decreases competition, and hydraulic shelter (Fausch, 1993). Cover can be measured at macro-, meso- and microhabitat scales (Allouche, 2002). Scientists that examine fish-habitat instream-flow relationships tend to use a microhabitat scale in their work, which gives them sufficient resolution of data to characterise fish habitat for management purposes (Vadas, 1992). However, many researchers have chosen to examine river fish abundance at the mesohabitat scale to get a broader impression of fish habitat preference (Vadas, 1992). The present study looked at the interactions between fish and these habitat factors at the mesohabitat scale, by comparing pool and riffle sites in terms of their fish assemblages and measures of physical habitat.

2.4.2: Field methods

When fish surveys had been completed at a site, a 100m tape measure was placed along the length of the habitat. In the pools, the measured area was from the outflow of the pool to the inflow, while in riffles it was the 20 meters of habitat between the block-nets. The pool was then subdivided into five transects, including one at each end of the habitat and three placed equidistantly between them. A 50m tape was placed along each transect, and habitat variables at four systematic points on each transect were measured. The first and last data-collection points on each transect were placed at 50cm from the edge, so as to take the cover features provided by the margins and wetted edge into account. The remaining two data collection points were placed at 1/3 and 2/3-channel width, thus making them equidistant relative to the banks and each other.

At each data collection point, the following habitat data were collected. Firstly, mean water column velocity was measured using a Scientific Instruments Mini-Kit flow meter and wading rod. The wading rod was also used to measure the depth (in cm) at the sampling point. In pools deeper than 1 m, no velocity was measured (as the flow meter was only designed to read velocity up to 1 m depth) and depth was measured using a marked pole. Next, the substratum particle lying on the point was identified and its embeddedness visually estimated as the percentage of its volume buried in the riverbed (after White & Harvey, 2001). The particle was then measured in millimetres across its beta axis (i.e. the second longest axis, perpendicular to the longest, alpha axis), which can be used as an approximation of mean diameter (Stream Systems Technology Centre, 1996). The amount of algae judged to be present on the particle was then noted on a qualitative scale of 0-5 (Table 4). Next, the metered pole was used to visually define a squared meter around the point, which on the first and last points of each transect extended from the wetted edge into the water. In this square meter, the percentage overhead cover, density of fine structures (defined as grasses, sedges and fine roots) and woody structures (defined as thick roots, branches, palmiet fronds and rhizomes) were estimated, as well as shading, on a scale of 0-5 (Table 4). The overall presence of detritus and coarse particulate organic matter (CPOM) in the habitat was also noted on a scale of 0-5 (Table 4). Physico-chemical measurements were also taken at each site, either during or directly after the fish survey. Temperature (°C) and pH were measured using a Hanna Waterproof pH meter, and conductivity was measured with a Crison 524 conductivity meter ($\mu S.cm^{-1}$). At four of the eight sites, maximum-minimum thermometers were left in the water at the start of the expedition and removed on the last day, so that the temperature range experienced in the river during the survey could be measured.

Table 4: Qualitative visual assessment categories for assessing food availability and shading at each site

Score	Algae (on substrate particle)	Detritus ($1m^2$ around sampling point)	CPOM ($1m^2$ around sampling point)	Shading ($1m^2$ around sampling point)
0	None present	None visible	None present	Direct sunlight
1	Thin layer (slick to touch)	Fine coating on rocks	Some scattered material	Some shade
2	Sparse green patches	Thick layer on substrate	Some leaf packs present	Partially shaded
3	Green over much of surface	Silt / flocculent matter combined	Many leaf packs present	Moderately shaded
4	Some clumped algae		Large accumulations of leaves	Mostly shaded
5	Thick algal mat			Complete cover

2.4.3: Seeking the influence of habitat availability on fish distributions

When attempting to explain the relationship between individual fishes and their habitat, it is crucial to understand the relationships that may exist between the habitat variables chosen for analysis. If two variables are linked in a direct or indirect relationship, then the influence they each impart on fish abundance cannot be considered to be independent. One can consequently never assume a cause-and effect relationship between fish and any one variable, as one is unsure as to what other factors are influencing both the fish and that individual variable. In order to gain insight into these potential relationships, or autocorreations, the untransformed means of all quantitative variables per site were placed in correlation matrices for pool and riffle sites using the computer package Statistica 6 (© Statsoft 2001). It was decided not to exclude any variables found to auto correlate, as this was would have imposed a value judgement on the importance of either variable to fish-habitat interactions.

The next step was to understand whether any of the habitat variables could provide an explanation for fish distributions. The study by Townsend & Crowl (1991), which investigated the relative impacts of alien trout and physical habitat on the distribution of native fish in New Zealand, utilised multivariate analyses that seemed appropriate to answer this question. In that paper, Discriminant Function Analyses (DFA) were used to determine whether habitat variables (shown previously through regression analyses to correlate strongly with fish abundance) could discriminate between sites that had only trout, sites with only indigenous fish, and sites containing both the indigenous and alien fish. The study was able to show that the presence of trout was a major factor dictating the presence or absence of galaxias at a site, and that the presence of physical barriers (such as waterfalls) was the most important variable discriminating between invaded and non-invaded sites (Townsend & Crowl, 1991).

While many exploratory multivariate analyses (e.g. Principle Component Analysis, Multi-Dimensional Scaling) are designed to find patterns and groups in a large data set, the key difference in utilising DFA is that it is used to find variables that classify groups defined prior to analysis (Quinn & Keough, 2002), and not to test whether these groups exist or not. For this study, The DFA was performed on all quantitative habitat variables collected, to see which would discriminate between four fish-habitat classifications of the sites (non-invaded riffle, non-invaded pool, bass-invaded riffle, bass-invaded pool). Since it was known from a pilot survey conducted in April 2003 that the riffles and pools were intrinsically different from each other in their physical characteristics, and since the invaded sites were known from that survey to be very different from non-invaded sites with respect to their fish diversity, these groups were thought to be real enough to justify the use of DFA rather than another exploratory multivariate technique. DFA is a parametric test, as it requires the data being analysed to be both homoscedastic and reasonably close to a normal distribution (Zar, 1999). In order to meet these requirements, all continuous variables were first log- or square-root-transformed, while percentage variables were arcsine-log transformed prior to running the DFA. Post-transformation verification of normality and homoscedasticity was performed in Statistica 6 prior to running the DFA. Variables that could still not be verified as normally distributed were discarded from the parametric analyses.

Linear regressions were next used to test for significant correlations between the untransformed habitat variables and fish densities. While multiple regression analysis as used by Townsend and Crowl (1991) would have been appropriate for this study, the sparseness of the fish data meant that key assumptions of multiple regression analyses (normality of distribution, a minimum of 20 site visits) could not be adhered to (Quinn & Keough, 2002). Since none of the fish abundance data were normally distributed post-transformation (Kolmogorov-Smirnoff test for normality, $p<0.05$), non-parametric Spearmann Rank Order Regressions (Zar, 1999) were utilised. If any habitat variables were shown by DFA to discriminate between invaded and non-invaded sites, these could now be investigated for their relationships with specific fish species densities. This could indicate whether or not these habitat variables played a significant role in the distribution of fish where bass were either present or absent. In order to remove the potentially confounding effect of the bass themselves on these relationships, only non-invaded pools and riffles were used for this analysis.

CHAPTER 3: RESULTS

3.1: FISH SURVEY RESULTS

3.1.1: Distribution of fish species in the Rondegat

Labeobarbus capensis, *B. calidus*, and *P. phlegethon* were recorded at pools and riffles surveyed upstream of the waterfall barrier during October, November and April, while *L. capensis* was the only indigenous fish recorded at bass-invaded sites (Tables 5 & 6). *Austroglanis gilli* was common at all non-invaded riffles, but was entirely absent at invaded riffles (Table 5). *Galaxias zebratus* were only found in the top two pool sites, in the mountain fynbos reach of the river (Table 6). The species is only known from the upper reaches of the river, its previously known range extending from the road bridge in the vicinity of Site 1 to about 1km upstream of Algeria (R. Bills, unpublished data, 2003).

Table 5: Numbers of fish captured by electrofishing at riffle sites in October 2003, November 2003 and April 2004. Data collected during 2nd passes in April are not used.

	Non-invaded					Bass-invaded					
October Riffles: Total Catch	Site 1	Site 2	Site 3	Site 4	**Total**	Site 5	no survey	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	0	1	9	0	**10**	0		0	0	0	**0**
Barbus calidus	1	0	4	14	**19**	0		0	0	0	**0**
Pseudobarbus phlegethon	0	0	7	6	**13**	0		0	0	0	**0**
Austroglanis gilli	15	18	38	28	**99**	0		0	0	0	**0**
Micropterus dolomieu	0	0	0	0	**0**	3		1	3	0	**7**
Lepomis macrochirus	0	0	0	0	**0**	0		0	0	2	**2**
November Riffles: Total Catch	Site 1	Site 2	Site 3	Site 4	**Total**	Site 5	Site B	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	0	0	10	4	**14**	0	0	1	0	0	**1**
Barbus calidus	1	0	22	12	**35**	0	0	0	0	0	**0**
Pseudobarbus phlegethon	6	2	22	17	**47**	0	0	0	0	0	**0**
Austroglanis gilli	14	18	32	22	**86**	0	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0	**0**	3	0	1	8	0	**12**
Lepomis macrochirus	0	0	0	0	**0**	0	0	0	0	3	**3**
April Riffles: Total Catch (1st pass only)	Site 1	Site 2	Site 3	Site 4	**Total**	Site 5	Site B	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	1	0	4	14	**19**	0	1	0	0	0	**1**
Barbus calidus	45	0	2	5	**52**	0	0	0	0	0	**0**
Pseudobarbus phlegethon	16	0	28	5	**49**	0	0	0	0	0	**0**
Austroglanis gilli	25	37	34	31	**127**	0	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0	**0**	7	0	0	0	0	**7**
Lepomis macrochirus	0	0	0	0	**0**	0	0	0	0	0	**0**

Table 6: Numbers of fish observed during snorkel surveys at pool sites in October 2003, November 2003 and April 2004. September data were generated from a single pass as the three-pass method had not yet been adopted.

	Non-invaded						**Bass-invaded**				
September Pools: Total seen (1 pass only)	Site 1	Site 2	Site 3	Site 4	no survey	**Total**	Site 5	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	5	0	0	1		**6**	14	13	2	0	**29**
Barbus calidus	2	3	21	42		**68**	0	0	0	0	**0**
Pseudobarbus phlegethon	0	0	0	1		**1**	0	0	0	0	**0**
Galaxias zebratus	0	0	0	0		**0**	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0		**0**	0	0	1	0	**1**
Lepomis macrochirus	0	0	0	0		**0**	0	0	0	0	**0**
October Pools: Rounded Means (3 passes)	Site 1	Site 2	Site 3	Site 4	no survey	**Total**	Site 5	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	2	1	38	11		**52**	12	8	1	0	**21**
Barbus calidus	226	75	67	82		**450**	0	0	0	0	**0**
Pseudobarbus phlegethon	8	1	20	8		**37**	0	0	0	0	**0**
Galaxias zebratus	0	2	0	0		**2**	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0		**0**	4	2	2	1	**9**
Lepomis macrochirus	0	0	0	0		**0**	0	0	0	0	**0**
November Pools: Rounded Means (3 passes)	Site 1	Site 2	Site 3	Site 4	no survey	**Total**	Site 5	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	17	4	38	28		**87**	47	1	0	0	**48**
Barbus calidus	330	60	71	114		**575**	0	0	0	0	**0**
Pseudobarbus phlegethon	45	1	15	15		**76**	0	0	0	0	**0**
Galaxias zebratus	2	2	0	0		**4**	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0		**0**	8	11	40	6	**65**
Lepomis macrochirus	0	0	0	0		**0**	0	0	0	3	**3**
April Pools: Rounded Means (3 passes)	Site 1	Site 2	Site 3	Site 4	Site A	**Total**	Site 5	Site 6	Site 7	Site 8	**Total**
Labeobarbus capensis	29	0	14	16	21	**80**	57	0	0	0	**57**
Barbus calidus	359	72	51	54	99	**635**	0	0	0	0	**0**
Pseudobarbus phlegethon	50	16	16	27	87	**196**	0	0	0	0	**0**
Galaxias zebratus	4	0	0	0	0	**4**	0	0	0	0	**0**
Micropterus dolomieu	0	0	0	0	0	**0**	1	0	0	0	**1**
Lepomis macrochirus	0	0	0	0	0	**0**	0	0	0	0	**0**

Barbus calidus was by far the most abundant species, with 1834 individuals counted in total (Figure 3A). *Micropterus dolomieu* were found at four of the five invaded sites (Tables 5 & 6). *Lepomis macrochirus* were recorded only at Site 8, and only eight were recorded in total during the surveys (Figure 4B). *Labeobarbus capensis* were recorded at invaded sites 5, B, 6 and 7 (Tables 5 & 6). River walks conducted along the invaded reach in October confirmed their presence between these sites, as well as downstream of Site 7. *Labeobarbus capensis* were never recorded at Site 8, and appear to be absent from the river downstream of an abstraction weir in the small canyon (Figure 1). This weir is a concrete structure shored up with sandbags (Appendix 3; Plate 13) and was approximately 0.5km upstream of the reservoir when full. The most significant result of the survey though, was that *B. calidus*, *P. phlegethon* and *A. gilli* were never recorded at bass-invaded study sites (Figure 3B).

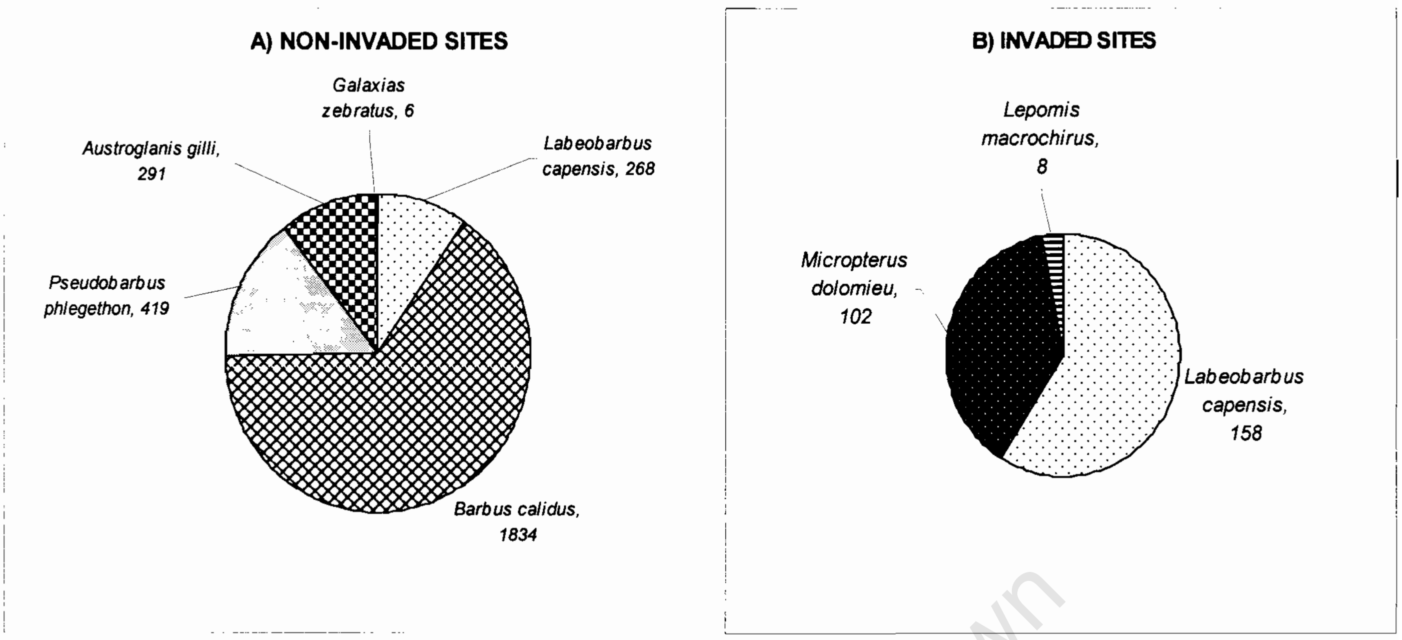

Figure 3: Combined survey counts for all species encountered at A) non-invaded sites and B) invaded sites in September 2003, October 2003 and April 2004.

3.1.2: Distributions of *Labeobarbus capensis*

A comparison of size classes recorded for *L. capensis* at non-invaded and invaded sites, with all field surveys combined, reveals a striking contrast (Figure 4). *Labeobarbus capensis* was recorded for all size classes except that of very large adults (>40cm) in non-invaded sites. This included age 0+ juveniles counted in the May 2003 pilot study and the April 2004 field trip, as well as small, medium and large juveniles and adults. The invaded sites, in comparison, show a near-complete absence of fish under 20cm. A single individual of 8cm was captured at Site B in April 2004. As this specimen was captured four days after a medium-sized flood was recorded on the river (D. Malherbe, CWA manager, pers. com. 2004), it may have recently been flushed downstream by the high flows. The four invaded sites, while having fewer individuals of *L. capensis* (Figure 3B), did have more yellowfish in the 20-30cm, 30-40cm and >40cm size classes, than were recorded upstream of the waterfall barrier. Relative densities recorded at three of these sites over the changing seasons appear to show a distinctive migration pattern in these large adult yellowfish.

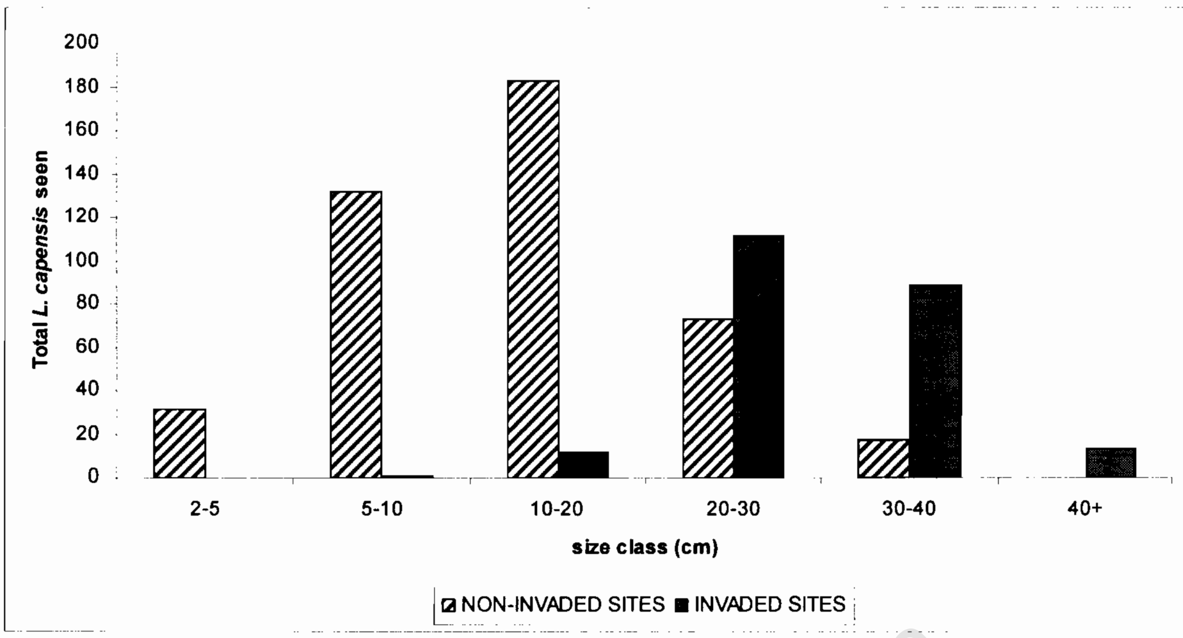

Figure 4: Length frequency distribution of *Labeobarbus capensis* at invaded and non-invaded sites, using combined data from surveys conducted in May, September, October, November 2003, and April 2004.

In the pilot study of May 2003, 50 individuals of *L. capensis* were found at the uppermost invaded pool site (Site 5), but were absent from the other downstream sites (Figure 5). Late winter rains caused the river to flood in early September 2003, and when the sites were visited in that month, individuals of *L. capensis* were found evenly distributed between all sites bar the lowest (Site 8). In October 2003, before any major spawning events appeared to have taken place, river walks confirmed that *L. capensis* was still evident in pools throughout the invaded river reach between the waterfall and the abstraction weir in the canyon.

In November, *L. capensis* was absent from the lower sites (apart from two individuals seen in the Site 6 pool) and the majority of the individuals appeared to have returned to the Site 5 pool, where 50 were recorded (Figure 5). A small shoal of larvae tentatively identified as *L. capensis* (SAIAB collection no. 75268) were captured in a sandbag weir pool approximately 1km downstream of the Site 5 pool, suggesting that the yellowfish had recently spawned nearby. In April 2004, no *L. capensis* were recorded anywhere in the lower river apart from in the Site 5 pool, their distribution now apparently mirroring the autumn distribution of the previous year.

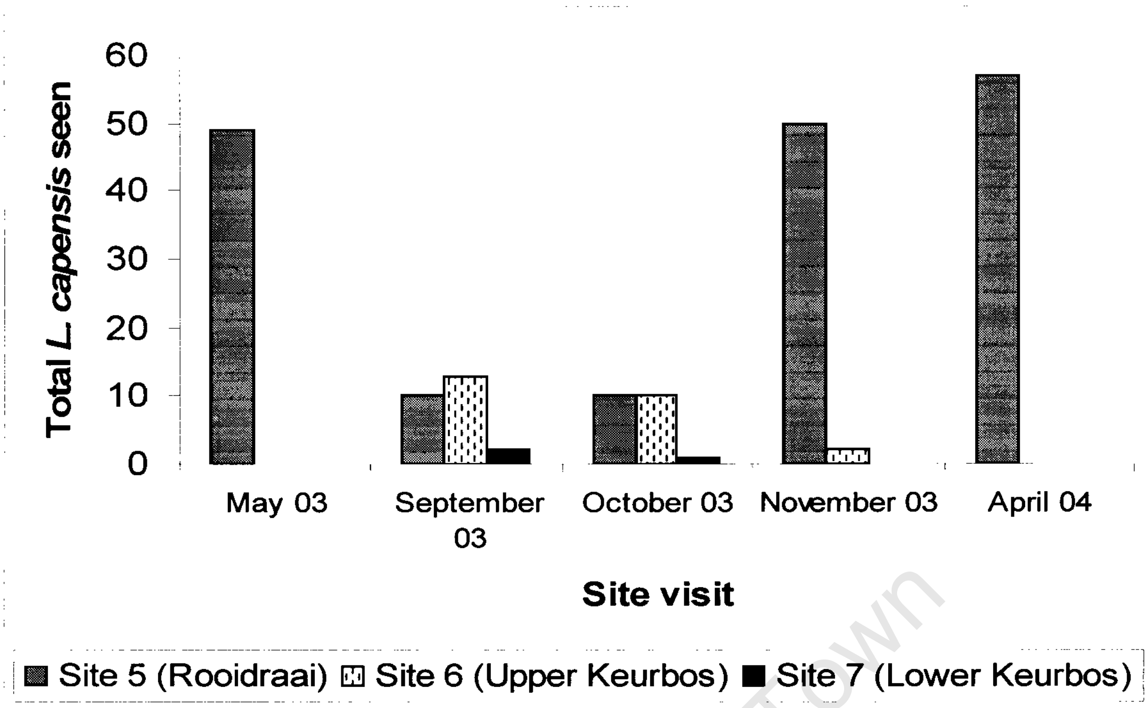

Figure 5: Comparison of seasonal abundances of adult *L. capensis* at three invaded sites

3.2: THE IMPORTANCE OF HABITAT IN FISH DISTRIBUTIONS

3.2.1: Interactions between habitat variables

The untransformed mean values of continuous and percentage variables generated at riffle (Table 7) and pool (Table 8) sites were placed in Spearman Rank Order correlation matrices in order to identify potential synergies between individual habitat variables. Both analyses detected a very close correlation between volume and depth. This is an expected result, as volume is simply a calculated function of depth and surface area. Significant negative correlations between volume and substratum size and embeddedness in riffles (Table 7) indicate that sand and silt were prevalent in larger, deeper riffle habitats. Substratum embeddedness also correlated positively with overhead cover, indicating a higher sediment load in riffles with thickly vegetated riparian zones.

Table 7: Correlation matrix of all untransformed continuous variables collected during habitat analysis in riffles. All significant ($p<0.05$) correlations marked in bold font

	Volume	Depth	Subst.	Emb.	Deposit Depth	Velocity	O.H. Cover	F. Struct.	W. Struct.
Volume	1.00	**0.76**	**-0.79**	**0.71**	-0.08	0.07	0.46	0.40	-0.09
Depth	**0.76**	1.00	-0.46	0.34	-0.41	0.49	0.32	-0.04	-0.30
Subst.	**-0.79**	-0.46	1.00	**-0.72**	0.12	0.24	-0.22	-0.44	-0.26
Emb %	**0.71**	0.34	**-0.72**	1.00	0.20	-0.23	**0.61**	0.48	-0.02
Deposit	-0.08	-0.41	0.12	0.20	1.00	-0.50	-0.11	**0.59**	-0.007
Velocity	0.07	0.49	0.24	-0.23	-0.50	1.00	-0.08	-0.52	**-0.58**
OH Cover	0.46	0.32	-0.22	**0.61**	-0.11	-0.08	1.00	0.18	-0.26
F. Struct.	0.40	-0.04	-0.44	0.48	**0.59**	-0.52	0.18	1.00	0.20
W. Struct.	-0.09	-0.30	-0.26	-0.02	-0.007	**-0.58**	-0.26	0.20	1.00

Woody structure density correlated positively with volume and negatively with velocity in pools (Table 8), indicating that the slower, larger pools tended to have larger amounts of woody debris. The positive correlation between water depth and sand deposit depth indicates that the deepest pools contained large sediment loads.

Table 8: Correlation matrix of all untransformed continuous variables collected during habitat analysis in pools. All significant ($p<0.05$) correlations marked in bold font

	Volume	Depth	Subst.	Emb.	Deposit Depth	Velocity	O.H. Cover	F. Struct	W. Struct.
Volume	1.00	**0.69**	-0.09	-0.01	0.25	-0.19	0.06	0.07	**0.70**
Depth	**0.69**	1.00	-0.18	0.47	**0.62**	-0.31	-0.33	0.10	0.35
Subst.	-0.09	-0.18	1.00	-0.35	0.11	-0.31	-0.54	0.33	0.03
Emb.	-0.01	0.47	-0.35	1.00	0.36	-0.29	-0.25	-0.38	-0.01
Deposit	0.25	**0.62**	0.11	0.36	1.00	-0.42	-0.49	0.002	0.06
Velocity	-0.19	-0.31	-0.31	-0.29	-0.42	1.00	**0.55**	-0.12	**-0.62**
OH Cover	0.06	-0.33	-0.54	-0.25	-0.49	**0.55**	1.00	-0.18	0.05
F. Struct.	0.07	0.109	0.33	-0.38	0.002	-0.12	-0.18	1.00	0.25
W. Struct.	**0.70**	0.35	0.03	-0.01	0.06	**-0.62**	0.05	0.25	1.00

3.2.2: Linking local abundance to habitat availability

(a) Discriminating between bass-invaded habitat and non-invaded habitat

Forward-stepwise discriminant function analysis (DFA) was performed on all transformed habitat variables. Six variables were found in descending order of importance to discriminate significantly between bass-invaded pools, invaded riffles, non-invaded pools and non-invaded riffles (Table 9).

Table 9: Habitat variables selected by forward-stepwise discriminant function analysis for a model to significantly discriminate between invaded pools, invaded riffles, non-invaded pools and non-invaded riffles ($p<0.05$).

Step	Variable	Wilks' Lambda	F-remove (3, 42)	p-level
1	log Volume	0.058	3.76	0.01
2	log √ Velocity	0.084	11.93	0.000009
3	log Depth	0.071	7.89	0.0002
4	√ Embeddedness	0.076	9.23	0.00008
5	log Deposit Depth	0.073	8.50	0.0001
6	log Woody Structure	0.049	1.16	0.33

Using the canonical analysis tool in Statistica 6, these discriminating variables were converted into three canonical factors, or roots. Graphic analysis of the spatial orientation of categorised sites along these roots revealed no discernable separation of categories when roots 2 and 3 or roots 1 and 3 were analysed. Graphic analysis of roots 1 and 2 showed clear separation of the four categories however, and only this analysis is shown here (Figure 6).

This graphic display shows that all riffles are separated from all pools by root 1 (X-axis). Factor analysis of the roots revealed volume, depth and the negative of velocity to be significantly correlated to root 1 (Table 10). This result makes intuitive sense, in that it indicates the riffles to be fast-shallow habitats, and the pools to be slow-deep habitats. The critical result of this analysis, however, is that invaded pools were separated from non-invaded pools along root 2 (Y-axis). There was also separation (though only partial) of invaded riffles from non-invaded riffles along root 2 (Figure 6). The factor structure matrix shows that the variables that best correlated with this root were habitat depth, substrate particle embeddedness and depth of sand deposits (Table 10). This correlation indicates that the invaded and non-invaded sites were separated along a gradient of depth and sedimentation, with invaded pools being deeper and having higher sediment loads than non-invaded pools. Likewise, at least some of the invaded riffles were significantly deeper and sandier than some of the non-invaded riffles.

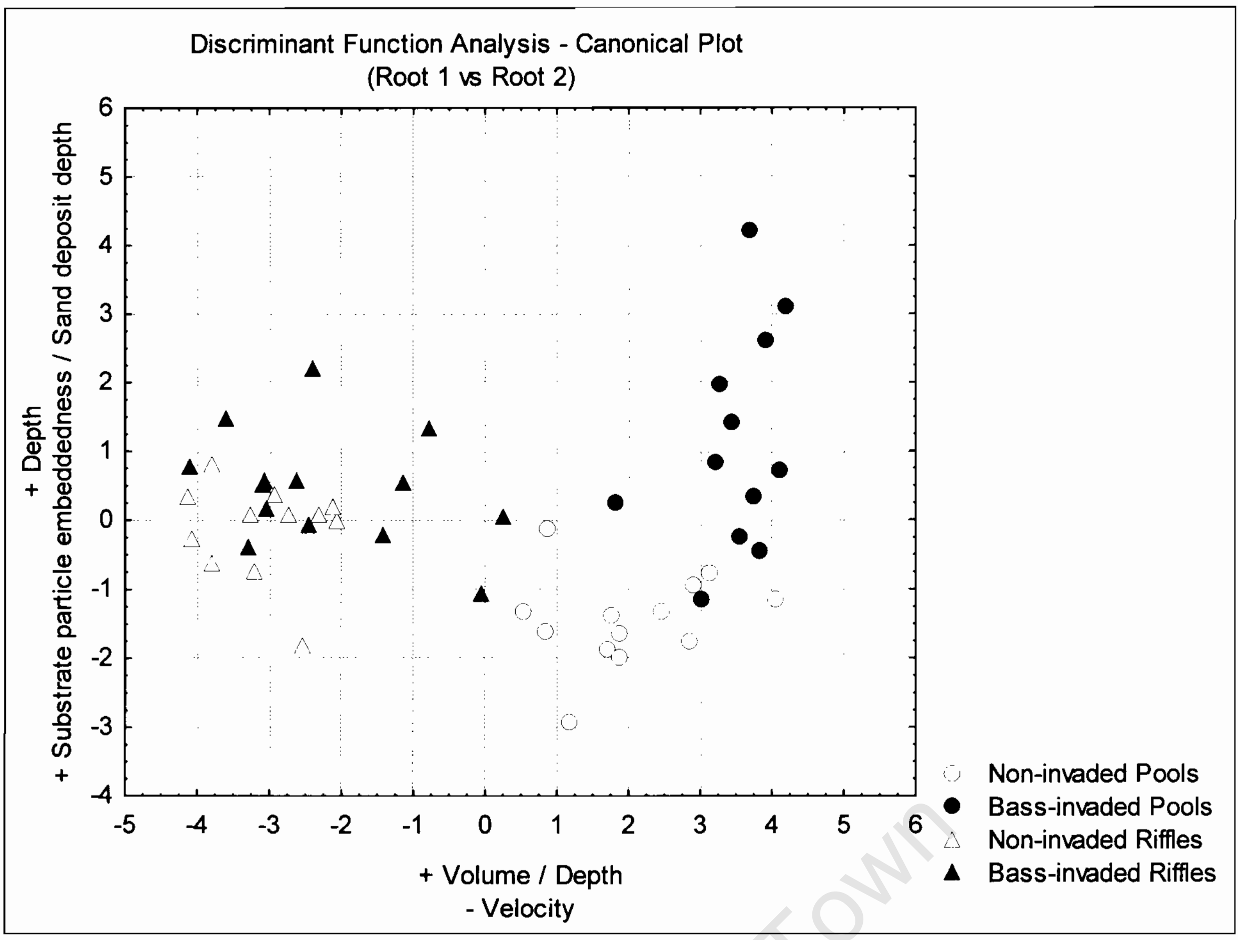

Figure 6: Canonical Plot of four fish-habitat categories analysed in DFA, indicating major habitat variables that comprise Root 1 (X-axis) and Root 2 (Y-axis).

Table 10: Factor structure matrix for canonical roots generated by DFA. Numbers indicate linear correlations of variables to canonical roots. Variables with correlation values in bold are also displayed on canonical graph axes (Figure 7).

	Root 1 r-value	Root 2 r-value
Log Volume	**0.68**	0.08
Log √ Velocity	**-0.44**	0.01
Log Depth	**0.67**	**0.45**
√ Embeddedness	0.15	**0.20**
Log Deposit	0.17	**-0.13**
Log woody structure	0.19	0.08

(b) Regressions of habitat variables against fish abundance at non-invaded sites

Non-parametric Spearman Rank Order correlation analyses were performed on untransformed species densities and all habitat variable means for non-invaded pools and riffles. For *L. capensis* and *B. calidus*, only pool survey data were used. In order to discern whether body size was a factor in the interactions of *L. capensis* and its environment, given the absence of small juveniles of the species at invaded sites, all *L. capensis* records were split into two size classes, namely small *L. capensis* (<10cm) and large *L. capensis* (≥10cm). For *A. gilli*, which was only detected using electrofishing, only riffle data were used. Separate analyses were performed for riffle and pool counts of *P. phlegethon*. Correlation analyses were also run between *M. dolomieu* and all habitat variables at bass-invaded pools and riffles.

Numbers of *B. calidus* and *P. phlegethon* in pools correlated positively with volume (Table 11), indicating that the abundance of these species increased proportionally with the size of the pool. Large *L. capensis* correlated positively with woody structure, suggesting a preference for instream structural cover. Small *L. capensis* correlated positively with overhead cover, but negatively with sand deposit depth (Table 11). This finding indicates the juveniles of *L. capensis* were less abundant at sites with heavy sediment loads.

Table 11: Spearman Rank Order correlations between species abundance and habitat variables in pools. Only significant (P<0.05) correlations are shown.

	n	Spearman Rank Order R	R^2	p
B. calidus & Volume	13	0.62	0.39	0.02
B. calidus & Depth	13	0.69	0.47	0.008
P. phlegethon & Volume	13	0.65	0.43	0.01
P. phlegethon & W. Struct	13	0.69	0.47	0.008
L. capensis (0-9) & Deposit	13	-0.55	0.31	0.04
L. capensis (0-9) & OH Cover	13	0.67	0.45	0.01
L. capensis (10+) & W. Struct	13	0.68	0.46	0.01

Abundances of *Austroglanis gilli* in riffles did not correlate significantly with any physical habitat variables, suggesting that the species was evenly distributed among the non-invaded riffle sites. *Pseudobarbus phlegethon* abundance correlated negatively with riffle velocity and positively with density of fine structure (Table 12). This indicates the species was most common at riffle sites with slower flows and large quantities of instream and marginal weeds and sedges. The important outcome of these analyses is that only small *L. capensis* among the indigenous species were counted in reduced densities at non-invaded sites with raised sediment levels.

Table 12: Spearman Rank Order correlations between species abundance and habitat variables in riffles. Only significant ($p<0.05$) correlations are shown.

	n	Spearman Rank Order R	R^2	p
P. phlegethon & Velocity	12	-0.73	0.54	0.006
P. phlegethon & F. Struct.	12	0.75	0.56	0.004

Micropterus dolomieu showed no significant correlations with any of the habitat variables in pools or riffles, suggesting that the species had no particular preference for any measured aspect of instream habitat. This could however be because the low densities of bass recorded at all sites made correlations hard to detect.

3.3: FEEDING BIOLOGY OF FISH IN THE RONDEGAT RIVER

3.3.1: Diet

(a) Volumetric analysis of the importance of food categories

The relative volumes of different food types found in the guts of the five species indicate that each species has its own particular food preferences. All five species relied heavily on aquatic invertebrates (Figure 7). The stomach contents of several individuals of *M. dolomieu* (Figure 8E), as well as a few of *A. gilli* (Figure 7D), consisted almost entirely of crabs (*Potamonautes* sp.), and so this taxon has been categorised separately.

A marked shift in the feeding preference of the three cyprinid species from the October and November (early-mid summer) samples to the early and late April (early-late autumn) samples was detected during analysis, and these seasons are displayed separately for all species in Figure 7.

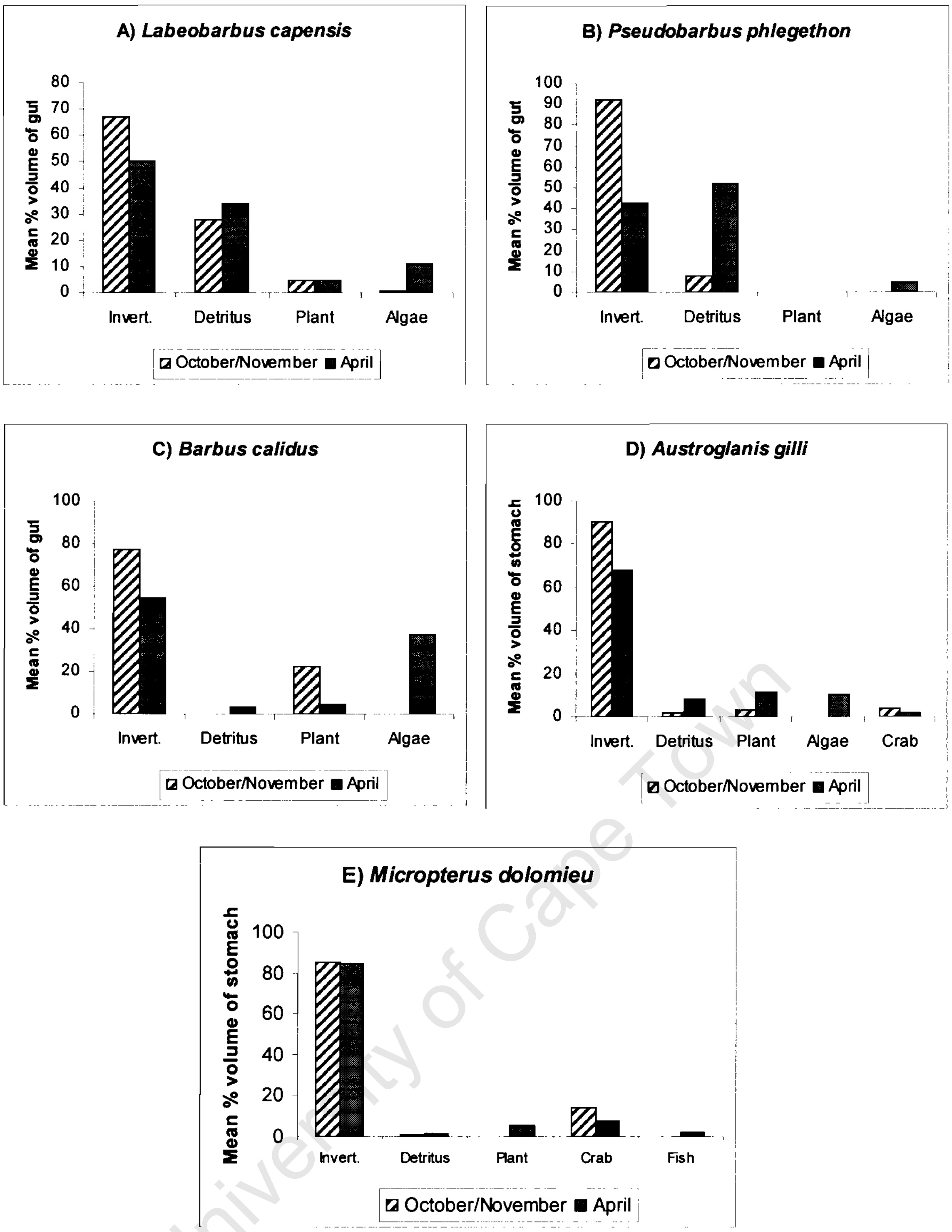

Figure 7: Mean proportional volumes of food categories found in stomach/gut analysis of five fish species in summer (October/November 2003) and autumn (April 2004). *Labeobarbus capensis*, gut n = 31; B) *Pseudobarbus phlegethon*, gut n = 30; C) *Barbus calidus*, gut n = 30; D) *Austroglanis gilli*, stomach n = 31; E) *Micropterus dolomieu*, stomach n = 32

While *P. phlegethon* (Figure 7B) fed almost exclusively on invertebrates in early summer, the individuals sampled in autumn fed extensively on detritus, as well as consuming some filamentous algae. *Barbus calidus* fed on invertebrates and some woody plant material (seeds, leaf and twig fragments) in summer, but fed on large quantities of filamentous algae in autumn (Figure 7C). During November field surveys, a shoal of *B. calidus* was observed feeding intensely on a patch of algae at Site 4, although filamentous algae did not become common at that site until autumn.

Labeobarbus capensis on average consumed similar proportions of invertebrates and detritus and its feeding preference did not shift seasonally (Figure 7A). Only three large (20cm+) individuals of *L. capensis* were collected for diet analysis, and all contained very large quantities of detritus (Appendix 2). This finding suggests non-selective feeding, with the adult yellowfish taking large indiscriminate mouthfuls of riverbed sediment, possibly in order to obtain unicellular organisms and benthic invertebrates. In contrast, *A. gilli* (Figure 7D) and *M. dolomieu* (Figure 7E) stomachs contained only small quantities of detritus, algae and plant material. This finding suggests that these species specifically target individual animals when feeding.

(b) Relative abundances of animal taxa in fish diets

All three cyprinid species, as well as *A. gilli*, fed heavily on benthic aquatic insect larvae of the orders Diptera, Ephemeroptera and Trichoptera (Figure 8). These data correspond to behaviour witnessed during snorkel surveys, where both *L. capensis* and *P. phlegethon* were observed feeding on and in between cobbles on the riverbed. The supposition that these species target the benthos as their primary source of food is supported by the significant quantities of benthic detritus (Figure 7) found in the guts of these species. *Barbus calidus* fed extensively on terrestrial invertebrate taxa (Figure 8C). These included various flying insects, as well as lepidopteran larvae. Many stomachs were found to have beetle fragments in them and, while some possibly belonged to terrestrial species, this could not be verified, so they have been represented separately from the other taxa in the pie-chart (Figure 8C).

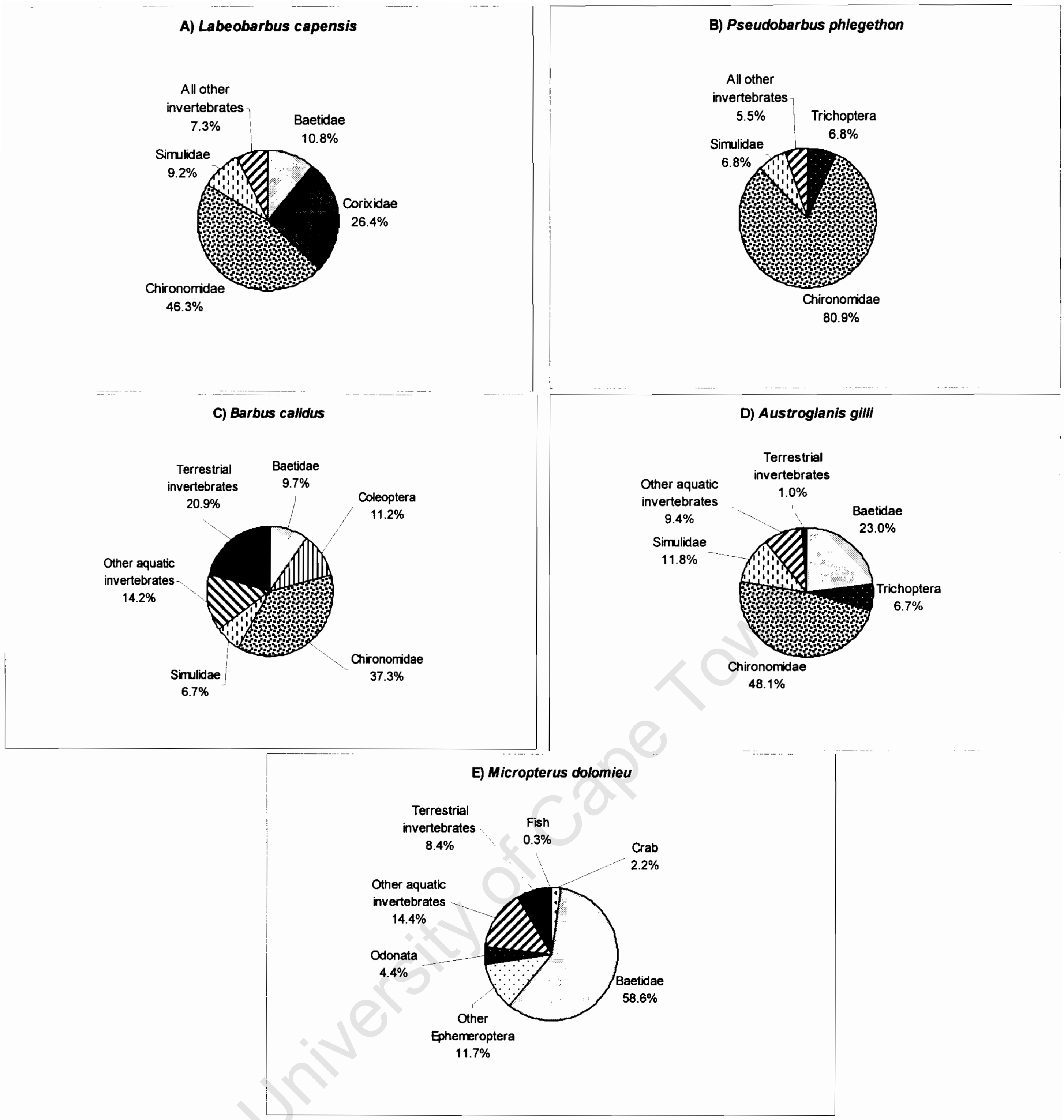

Figure 8: Proportional abundance of dominant animal taxa found in stomach/gut analysis of four fish species. A) *Labeobarbus capensis*, gut n = 31; B) *Pseudobarbus phlegethon*, gut n = 30; C) *Barbus calidus*, gut n = 30; D) *Austroglanis gilli*, stomach n = 31; E) *Micropterus dolomieu*, stomach n = 32

When one considers that some of the plant matter (leaves, seeds) found in *B. calidus* stomachs (Figure 7) was probably misidentified food taken off the surface, the data suggest that *B. calidus*, especially in early summer, was targeting the surface and water column for drifting invertebrates more than the benthos.

Micropterus dolomieu also fed extensively on terrestrial invertebrates (Figure 8E), while large specimens were often found to contain a single large crab (Appendix 2). The only fish discovered in a bass stomach was 14mm long and had lost most of its diagnosic features. Its overall shape, and the presence of a fully formed 'true' stomach, ruled out the family Cyprinidae. This meant it had to be either *A. gilli* or *M. dolomieu*. Since it lacked the fin spines diagnostic of most life history stages of Austroglanidids, it was most probably a young-of-the-year bass.

3.3.2: Availability of invertebrates as food

In order to clearly contrast the seasonality of invertebrate diversity and distributions, the October and April datasets of benthic kick sampling and drift net sampling are represented in Tables 13 and 14. While a great number of taxa was collected, only those that were found in fish guts are shown.

The majority of aquatic invertebrates found in the benthos were at some stage also recorded in the drift samples, suggesting most of these invertebrates spend some of their time drifting or swimming in open water, though fish could also obtain them by actively feeding on the bottom. Some invertebrate families, such as Heptageniidae, were found in both invaded and non-invaded sites all year round in the benthos, but were entirely absent from the drift. Other families, such as the Baetidae, were common all year round throughout the river in both the benthos and the drift (Tables 13 & 14).

Members of the Chironomidae, a dipteran family that all four indigenous fish species fed on heavily (Figure 8), were common throughout the river in the benthos, but were less common in the drift, except in April, which was possibly a flow-related anomaly. It is likely that fish feeding on chironomids were generally feeding off the benthos and not in the drift.

Table 13: Semi-quantitative abundances for key families of aquatic invertebrates collected by kick sampling at four representative sites on the Rondegat River during spring and autumn surveys. 1 = 1 animal; + = 2-9 animals; ++ = 10-100 animals

	Non-invaded		**Invaded**	
	Site 1 October	**Site 3 October**	**Site 5 October**	**Site 7 October**
Potamonautidae	1	++		
Baetidae	+		++	++
Caenidae			+	+
Heptageniidae	+	+	+	+
Leptophlebiidae	1	1	+	+
Coenagrionidae	1	1	+	+
Aeshnidae			1	
Gomphidae		+		+
Libellulidae		1		+
Corixidae				
Hydropsychidae	+	1	+	+
Leptoceridae	+	++	+	+
Dytiscidae				1
Gyrinidae	1	1	+	+
Chironomidae	+	+	+	+
Simulidae	+	+	+	++
	Site 1 April	**Site 3 April**	**Site 5 April**	**Site 7 April**
Potamonautidae			1	+
Baetidae	+	++	+	++
Caenidae			1	1
Heptageniidae	+	+	+	
Leptophlebiidae	+			
Coenagrionidae	+	+	+	+
Aeshnidae		1	1	+
Gomphidae		1	+	+
Libellulidae	1	+	+	+
Corixidae		1	++	+
Hydropsychidae		+		+
Leptoceridae				1
Dytiscidae				
Gyrinidae				1
Chironomidae		+	+	+
Simulidae	+	++	+	++

The general pattern that emerges from these results is that while some taxa were seasonally absent from either the drift or the benthos, all taxa targeted by fish were at some stage recorded both at bass-invaded and non-invaded sites. This in essence means that these taxa should not be a limiting factor preventing fish from utilising the lower river.

Table 14: Semi-quantitative abundances for key families of aquatic invertebrates collected in drift net at four representative sites on the Rondegat River during spring and autumn surveys. 1 = 1 animal; + = 2-9 animals; ++ = 10-100 animals; +++ = >100 animals

	Non-invaded		Invaded	
	Site 1 October	**Site 3 October**	**Site 5 October**	**Site 7 October**
Baetidae	+	1	+	+
Caenidae		+	++	1
Leptophlebiidae		+	+	
Aeshnidae	+			
Gomphidae	+	+	1	
Libellulidae	1	+	+	
Corixidae				++
Hydropsychidae	1			+
Leptoceridae		+	+	
Dytiscidae				
Gyrinidae				+
Chironomidae	1			1
Simulidae				
Terrestrial Coleoptera	++	+		
Terrestrial Hemiptera	1	+		
Terrestrial Diptera		+	+	
Other Terrestrial Invert.s	++	+	+	+
	Site 1 April	**Site 3 April**	**Site 5 April**	**Site 7 April**
Baetidae	++	+	+	++
Caenidae				
Leptophlebiidae	1			
Aeshnidae	+			
Gomphidae				++
Libellulidae		1		++
Corixidae	++	++	++	+++
Hydropsychidae				+
Leptoceridae	1			+
Dytiscidae	1			+
Gyrinidae				
Chironomidae	+	1	1	+
Simulidae	+	1		++
Terrestrial Coleoptera	+			+
Terrestrial Hemiptera		+		+
Terrestrial Diptera	+	+		
Other Terrestrial Inverts	+++	+	+	+

3.3.3: Availability of alternative food sources

Qualitative categorical scores for algae, detritus and coarse particulate organic matter (CPOM) were recorded at each data point during habitat analysis. These were assessed after collection and the highest scores recorded within each site were summarised to give an overall impression of relative abundances of food types. The September field trip is not included in this analysis, as the definitions used by this study to discern algae from detritus changed between the September and October trips.

The highest scores for CPOM (3 – Many leaf packs; 4 – Large leaf accumulations) were recorded at Sites 3, 4, 5 and 8 during all surveys (Table 15). All four sites had alien-tree-infested riparian zones, suggesting that the allocthonous material input had been significantly increased by the presence of both *Acacia mearnsii* (at Sites 3, 4 and 5) and *A. melanoxylon* (at Site 8) in the riparian zone.

There was a consistent contrast between detritus levels in riffles and pools during all seasons, in that category 3 (Silt/flocculent matter) was recorded in most pools, but very rarely in riffles (Table 16). This is clearly because the higher flows in riffles prevented the settling of silt and thick layers of detritus.

Table 15: Highest categorical abundance scores recorded for coarse particulate organic matter (CPOM) at bass-invaded and non-invaded sites in October 2003, November 2003 and April 2004. Definitions of scores are shown in Table 4.

	October		November		April	
Non-invaded	Pool	Riffle	Pool	Riffle	Pool	Riffle
1. Algeria	2	2	0	1	0	0
2. Rockface	1	1	2	1	1	1
3. Meadow	1	3	2	2	2	2
4. Cottage	3	1	3	1	3	1
Bass-invaded						
5. Rooidraai	4	3	4	2	4	1
6. Upper K	2	1	2	1	0	0
7. Lower K	2	0	0	2	1	0
8. Canyon	4	1	4	1	4	3

Table 16: Highest categorical abundance scores recorded for detritus at bass-invaded and non-invaded sites in October 2003, November 2003 and April 2004. Definitions of scores are shown in Table 4.

	October		November		April	
Non-invaded	Pool	Riffle	Pool	Riffle	Pool	Riffle
1. Algeria	2	2	3	1	2	0
2. Rockface	3	1	3	0	2	0
3. Meadow	2	1	3	1	3	2
4. Cottage	3	0	3	1	3	2
Bass-invaded						
5. Rooidraai	3	1	3	3	3	2
6. Upper K	3	0	3	0	3	0
7. Lower K	3	2	3	1	3	3
8. Canyon	3	0	3	0	3	1

The pools, especially in the lower river, had substrata that were often coated with a thick layer of fine silt and organic matter. This material was easily disturbed by the diver during snorkel surveys, and consequently had a tendency to affect visibility on the final pass of some of the pool surveys. It is interesting to note that at Sites 3 and 4, detritus increased significantly in the riffles between November and April (Table 16). This could signify the build-up of diatoms and other unicellular organisms in the benthos with the low flows at the end of summer.

The highest recorded scores for algae per site indicate that algae were overall more abundant in non-invaded sites than in bass-invaded sites all year round, while being particularly prolific at Sites 3 and 4 in April, where scores of 5 (thick algal mat) were recorded (Table 17). Dense mats of filamentous algae were abundant at these sites during collecting trips in February and April, while they were never recorded at the other sites at any time. As with the detritus at these sites, the algae were able to increase in abundance over weeks of low flow and warm, sunny conditions.

Table 17: Maximum categorical abundance scores recorded for algae at bass-invaded and non-invaded sites in October 2003, November 2003 and April 2004. Definitions of scores are shown in Table 4.

	October		November		April	
Non-invaded	Pool	Riffle	Pool	Riffle	Pool	Riffle
1. Algeria	2	3	2	3	3	3
2. Rockface	2	3	3	3	3	2
3. Meadow	2	2	3	3	3	5
4. Cottage	3	3	3	3	3	5
Bass-invaded						
5. Rooidraai	3	2	2	2	3	2
6. Upper K	2	2	2	2	2	2
7. Lower K	2	2	2	2	1	2
8. Canyon	3	2	2	1	4	2

CHAPTER 4: DISCUSSION

4.1: THE DISTRIBUTION OF FISH SPECIES WITHIN THE RONDEGAT RIVER

From the snorkelling and electrofishing surveys conducted on the Rondegat River from September 2003 to April 2004, it is clear that the fish community has become polarised, with an apparently near-natural population present above the barrier waterfall at Rooidraai, and what seems to be a highly impacted community below the barrier. At the lower sites invaded by *M. dolomieu*, four of the indigenous species, *A. gilli*, *P. phlegethon*, *B. calidus* and *G. zebratus*, were wholly absent (Figure 4). While *L. capensis* was found in bass-invaded sites, it was in significantly lower numbers than downstream, and all but one of these fish were over 10 cm in length (Figure 4). This finding suggests a complete absence of juvenile recruits in the lower river, and that recruitment is the result of immigration of adults that pass over the waterfall.

This is not the first study in the Western Cape that has found significant losses of indigenous fish diversity at sites that also contain *M. dolomieu*. On a field survey conducted in the Olifants-Doring system with Prof. P. Skelton, Gaigher (1973) found the upper reaches of the Ratels River to have "literally an aquarium of redfins", while the river downstream of a natural barrier was "barren except for an occasional undersized smallmouth bass". Christie (2002) found no specimens of two indigenous fish species, *P. burchelli* and *S. capensis*, as well as a near-total absence of the large cyprinid *Barbus andrewi* (Barnard) in invaded reaches of the Hex River in the Breede River system. All three species were found in abundance upstream of a causeway that served as a barrier to the upstream movement of *M. dolomieu*. Similarly, on the Witte River, another tributary of the Breede River, *P. burchelli* and *S. capensis* were found in very low numbers in a reach of the river invaded by *M. dolomieu*, while being abundant upstream of a bedrock shelf that formed a barrier to bass movement (Shelton, 2003).

In order to demonstrate convincingly the impact of *M. dolomieu* on the Rondegat River, the critical question to answer is which indigenous fish absences are due to natural distributions. Unlike in some European river systems, where most species are known to inhabit finite longitudinal zones within the river (Wooton, 1990), very little is known about the natural zonation of indigenous fish in the CFR, except for anecdotal evidence and historical records that suggest that some species were more common in the mainstems of the river than in the tributaries (Skelton, 1987). For instance, the apparently narrow range of *G. zebratus*, which was only ever seen at the uppermost two sites within the confines of the Cedarberg Wilderness Area, suggests that this species was either insufficiently sampled, or otherwise could be a mountain stream specialist that has never naturally occurred in the lower river. The field surveys of Bills (1999) found the species is at its most abundant upstream of the Algeria campsite, well above the study area for this project. Bearing this in mind, I assume that *G. zebratus* is not a suitable indicator of bass impacts, and so the remaining four indigenous species are the focus of the rest of the discussion.

It is known that *B. calidus* and *P. phlegethon* were native to the Olifants River mainstem, since the type localities of these species are to be found there (de Moor & Bruton, 1988). When Dr R.A. Jubb visited the mainstem as early as 1958 and 1959, these species had already vanished from the river (Gaigher, 1973). *Austroglanis gilli* is still found in small numbers in the mainstem of the Olifants River, as Bills (1999) captured some individuals downstream of Citrusdal, as well as below Clanwilliam dam. Based on this knowledge alone, one might have expected to find these three species in the lower Rondegat River. There is evidence that at least one indigenous fish species has already disappeared from the invaded reach of the Rondegat River. Van Rensburg (1964) captured several large specimens of Clanwilliam sawfin (*B. serra*), in the lower reaches of the Rondegat, but commented that there were no juveniles present. When Bills (1999) surveyed the Rondegat River, *B. serra* was not found.

4.2: LINKING FISH DISTRIBUTIONS TO ASPECTS OF PHYSICAL HABITAT

4.2.1. Data issues and constraints

When one looks at a small system in isolation, it is often difficult to separate out the interacting habitat variables that have contributed to the present distribution of a fish species. This is because if two causal factors result in the same outcome, it may be impossible to decide which factor was more important. In this case, the absence of a fish species from a certain habitat is often not the result of only one ecological constraint, but rather a combination of constraints that may either act in synergy or mitigate against each other in dictating how well a fish will cope with its environment. Ideally, if one wanted to identify habitat factors that fish were interacting with (e.g. the presence of benthic cover for predator avoidance), one would want to test the relative importance of this factor through comparative analysis of several rivers where it was perceived either to be important (e.g. cobble-bed streams) or to be non-existent (e.g. sand-bed streams). This approach would create a data set of independent scenarios of fish-habitat interactions that could be used to validate the presence or absence of the interaction on the Rondegat River.

Such an approach is not possible in the present case, because funding, time and biological constrains limited the scope of the project to the Rondegat River. In this case, by utilising multivariate exploratory statistics, it was possible to isolate from the many variables measured those that best correlated with fish distribution patterns, and could therefore suggest cause-and-effect relationships. This approach was preferable to choosing one potential process *a priori* and studying it to the exclusion of other potentially important processes, in that it would be more informative from a management perspective.

In any research that seeks to detect fish-habitat interactions, there is a fundamental issue of getting sufficient replicates. Some studies seeking to identify the physical habitat factors controlling fish distributions have utilised over 100 sites for their analyses (Townsend & Crowl, 1992; Martin-Smith, 1998). Since the indigenous fish of the Olifants-Doring system are so scarce, so restricted and their habitats often so hard to access, one cannot emulate the methods used by researchers in circumstances where fish are more common and accessible. In the case of the Rondegat, replicates were created through the revisiting of each site during the four seasonal surveys. This technique, which could be seen as a form of pseudo-replication (Stewart-Oaten *et al.*, 1986), is nonetheless a valid approach in the case of freshwater fish, which are often seasonally variable in their local distributions. Pires *et al.* (1999) used this technique to detect the physical habitat factors that dictated seasonal distributions of fish within intermittent streams in Portugal.

The other challenge that small data sets such as this one present is the availability of appropriate statistical techniques. The small number of replicates, coupled with the low densities of fish, meant that parametric multiple regression techniques generally used to investigate the links between fish and habitat (Townsend & Crowl, 1991; Walters & Wilson, 1996; McIntosh, 2000) could not be used. Instead, non-parametric Spearman Rank Order Regression was utilised. The results gained from this analysis are still valid however, as long as one recognises the limitations of the applicability of the test. A significant correlation between the local abundance of a species and a habitat variable detected by this test can be considered meaningful in-as-much as it pertains only to the sites from which the data were gathered, and cannot be assumed to take place in other areas where the species may occur (T. Dunne, HOD Department of Statistics, UCT, pers. com. 2005). Thus, the fact that a species correlated strongly with a habitat variable in the upper river is no guarantee that it would react in the same way in the lower river, if that habitat were in any way different from the sites where the information was gathered.

4.2.2: The influence of habitat quality on fish distributions

Discriminant function analysis (DFA) of the various habitat variables demonstrated that water depth and two aspects of sedimentation were the factors best able to discriminate between bass-invaded and non-invaded pools (Figure 6). These aspects of sedimentation were the mean embeddedness in the riverbed of individual substratum particles, and the mean depth of sand deposits recorded at each sampling point. These variables also partially discriminated between bass-invaded and non-invaded riffles. What these findings indicate is that if physical habitat quality were an alternative limiting factor (other than bass) affecting the distribution of fish species within the Rondegat River, then sedimentation would be the most likely disturbance factor that prevented indigenous species from inhabiting the lower reaches.

Siltation from bank erosion has long been seen as a major threat to fish communities in the United States, where the chief impacts on fish are listed as physical alteration of stream habitat and decreased survival of eggs and larvae (Walser & Bart, 1999). It also clogs respiratory organs, reduces the foraging success of aquatic organisms and disrupts the food web of the stream (Belsky *et al.*, 1999). Sedimentation was mentioned by Skelton (1987) and Gaigher *et al.* (1980) as being a potential threat to indigenous fishes in the CFR. The most likely effects of sedimentation on fish in Cape rivers are loss of spawning habitat and potentially of food through the negative impacts of siltation on benthic invertebrates (Skelton, 1987; Gaigher *et al.*, 1980). Bills (1999) specifically mentioned sedimentation as a threat to *A. gilli*, in that sediments filled in the cobble interstices in which they hide, making them more vulnerable to diurnal predators.

When considering sedimentation as a form of disturbance, it must be remembered that most rivers tend to experience increased sedimentation as they flow downstream, as shallowing gradients lead to increased deposition of sediment from the water column (Davies & Day, 1998). One might therefore expect the lower sites of the Rondegat to be sandier than the upper, but when one considers that the gradient of the river did not significantly decrease between the waterfall barrier and the Clanwilliam reservoir (Figure 2), the large amounts of sediment found in the invaded pools were more likely caused by disturbance of the river channel upstream.

Linear regressions run between indigenous fish abundance and localised aspects of physical habitat, showed none of *A. gilli*, *B. calidus* or *P. phlegethon* to be negatively correlated with embeddedness or sand deposit depth. This suggests that none of these species should be less common in the lower river, provided that other aspects of the instream habitat do not differ significantly from the upper river where the analysis was carried out. Small (0-9cm) *L. capensis* did show a negative correlation with sand deposit depth in pools (Table 11). While this result could suggest avoidance of sandy habitats by *L. capensis*, it must be noted that this correlation was barely significant from a statistical point of view ($p = 0.04$) and might have been generated by a random distribution of the juvenile fish between the sites. The likelihood of the Spearman Rank Order Regression producing a falsely significant result could not be tested as it is a non-parametric regression (Zar, 1999), but the possibility of such an error cannot be ignored. This result does not therefore offer a serious alternative explanation to the apparent loss of juvenile *L. capensis* in the lower river, especially since it cannot explain the near-complete absence of *L. capensis* juveniles in bass-invaded riffles. Small *L. capensis*, while rarer than the other species, were consistently caught at all non-invaded riffle sites.

In an attempt to gauge the potential of the upper Rondegat for invasion by bass, regressions were performed between the site-specific densities of *M. dolomieu* and the habitat variables recorded at invaded sites. As with *A. gilli*, no habitat variables were found to correlate significantly with densities of *M. dolomieu*. This may indicate one of two things. The first is that bass densities in the lower Rondegat were too low for linear regressions to be able to detect trends in abundance. The other possibility is that *M. dolomieu* is so robust in its habitat requirements that differences in habitat quality between the lower sites had no impact on local densities.

Bass were found in relatively small numbers (generally less than ten per site) and more individuals of all sizes were counted in pools than in riffles. Juvenile *M. dolomieu* are riffle-run specialists in many North American rivers, while large adults of the species are more common in pools (Walters & Wilson, 1996). This segregation is the result of the juveniles' dependency on the sediment-free cobble substrata of riffles for food and cover, as well as their avoidance of adult *M. dolomieu*, which prey on them (Walters & Wilson, 1996). The low recorded densities of bass in riffles in this study could be due to under-sampling, as centrarchid fish are known to be difficult to sample with electrofishing, especially when there is instream cover for them to hide in (Meador *et al.*, 2003).

If bass were to penetrate upstream of the waterfall barrier, it is likely that juvenile bass would rapidly move upstream to avoid adults established downstream and, given the favourable physical habitat available in the upper reaches, these bass would quickly invade the remainder of the Rondegat River. This pattern of rapid invasion has been witnessed in many other rivers throughout the Western Cape (de Moor & Bruton, 1988).

4.3: LINKING FISH DISTRIBUTIONS TO FEEDING AND BREEDING BIOLOGY

4.3.1: Biology of *Barbus calidus* and *Pseudobarbus phlegethon*

Since the absence of *B. calidus* and *P. phlegethon* in the lower river cannot be readily explained by differences in physical habitat quality to that at upstream sites, the two remaining possibilities left to explore are the availability of food and the presence of *M. dolomieu*.

Dietary analysis showed both redfin minnow species to be omnivorous, targeting invertebrates as well as benthic detritus and filamentous algae. While *B. calidus* appeared to prefer algae to detritus, *P. phlegethon* fed extensively on detritus, rather than algae (Figure 7). When the availability of detritus and algae were estimated during field surveys, it was found that detritus was abundant throughout the river (Table 16), while algae tended to be more abundant at bass-free sites (Table 17). This suggests that if food really were a limiting factor in the lower river, then *B. calidus* would have been less common in the lower Rondegat than *P. phlegethon* before the invasion of bass. However, since both species fed mostly on invertebrates, which were common throughout the river both in spring and autumn (Tables 13 & 14), and since algae and detritus only became abundant in late summer, it is doubtful that food availability has ever limited the distribution of either species. An alternative possibility is that the lower river lacks sufficient breeding habitat for *B. calidus* and *P. phlegethon.*

Barbus calidus is known to spawn in deep runs and pools where they deposit eggs under boulders and in bedrock cracks (Impson & Swarz, 2002). These observations suggest a potential lack of spawning ground for *B. calidus* in the lower Rondegat River, where some study sites were dominated by sandy substrata and very little bedrock. Bills (1999) found *B. calidus* to be more common in the upper and lower reaches of tributaries, where the river is characterised by deeper pools and bedrock/boulder substrata, than in the middle reaches that were characterised by shallow, cobble-dominated substrata. This could suggest spawning habitat to be a limiting factor in the lower Rondegat River, particularly in the area of Keurbos Farm where study sites were dominated by sandy substrata and very little bedrock. Even so, immigration from upstream of the waterfall barrier ought to sustain a population of *B. calidus* in the lower river, even if those fish seldom breed successfully. A potential lack of spawning habitat cannot explain the species' complete absence below the waterfall.

Pseudobarbus phlegethon is known to spawn over open cobble substrata, the eggs settling on the surfaces of the cobbles (R. Bills, pers. com. 2004). Bills (1999) observed *P. phlegethon* to be most common in the middle reaches of Olifants River tributaries, where complex cobble substrata were most abundant. From these findings, it seems that *P. phlegethon* may have been more common than *B. calidus* in the lower-middle reaches of the Rondegat River before the invasion of bass, and that there is still likely to be sufficient spawning habitat for this species in the riffles of the invaded reach, which are not as badly affected by sedimentation as the pools are (Figure 6).

4.3.2: Biology of *Labeobarbus capensis*

Labeobarbus capensis was the only indigenous species to be recorded at bass-invaded study sites, where smaller numbers of adults, and very few sub-adults were recorded compared to upstream sites. The idea that food availability could have limited *L. capensis* abundance in the lower river is ruled out by the species' feeding ecology. Yellowfish of all sizes fed mostly on invertebrates (Figure 7), particularly on members of the families Chironomidae and Corixidae (Figure 8), both of which were as common (or more common) in the benthos and drift of the invaded sites as in the non-invaded sites (Tables 13 & 14).

An alternative argument that could explain the abundance and size class ratio patterns of *L. capensis* (Figure 4) is that it is a result of spawning migrations. Based on anecdotal accounts, the prevailing perception of *L. capensis* is that it once migrated up the Olifants River mainstem in summer to spawn in the tributaries (Cambray *et al.*, 1997), and that the juveniles over many seasons migrated downstream back to the mainstem as they grew older. The tendency of juveniles to be most abundant in mountain streams and foothills, and large adults more common in lower reaches, supports this perception (Skelton, 1987). If this information is correct, it could be argued that the adult yellowfish in the lower Rondegat River were migrating up past the waterfall barrier in spring and spawning along with all the other mature yellowfish in the upper river, before migrating back down to the lower river to live out the rest of the season there, thereby explaining the lack of juveniles in the invaded sites.

However, the migratory patterns of *L. capensis* recorded at the lower sites over the course of the project appeared to be very different from the above scenario. The yellowfish in the invaded section appeared to behave as a separate sub-population cut off from the rest of the yellowfish upstream, and followed a spawning migration linked to both flow and temperature cues. At the end of summer in 2003 and 2004, the water level was very low, and the majority of adult yellowfish appeared to have accumulated in the Site 5 pool. This is the biggest and deepest pool encountered in the entire Rondegat River, making it an attractive refuge for large fish in a river experiencing low flows. In September 2003, sustained winter floods allowed the fish to migrate downstream, distributing themselves evenly between the study sites (Figure 6), as well as downstream towards the small canyon, although they appear not to have penetrated past the abstraction weir situated there (Figure 1). This migration was probably partly passive, as the winter floods created violent flows, even in deep pools (pers. obs., 2003), and partly active, in that the fish would have needed to seek out new feeding opportunities from fresh habitats after spending the majority of the dry season trapped together in a confined pool.

The yellowfish were still present throughout the river in October 2003, but between that survey and the following one, at the end of November, the majority of yellowfish appeared to return *en masse* upstream to the Site 5 pool. Most significantly of all, it was during the November survey that larvae believed to be *L. capensis* were spotted 1km downstream of the Site 5 pool. A mass spawning migration seems to have occurred, with all the yellowfish moving upstream to breed in spawning beds near to the Rooidraai pool, after which they returned to the pool's protective depths where they 'wait out' the rest of the summer. The spawning beds are likely to have been shallow runs and riffles with a gravel-cobble-dominated substratum, which have been identified as the preferred substrata in the mainstem of the Olifants River (King *et al.*, 1998). Migration was probably triggered by temperature and water velocity cues, which would have informed the yellowfish that the spawning beds were in good condition to allow a successful spawning event (B. Paxton, UCT, pers. com. 2004). This pattern of timed migration and spawning within a tributary has been observed in Clanwilliam sawfin (*B. serra*) in the nearby Driehoeks River (B. Paxton, unpublished data).

The implication of these findings is that *L. capensis* individuals were spawning in the bass-invaded lower reaches of the Rondegat River during the summer of 2003/2004, and may have done so for many years, but there has clearly been close to no recruitment of juveniles following those spawning events, even though it appears that some of the eggs hatched into larvae. The best explanation left to account for this scenario is that recruitment to the adult population is the result of immigration from upstream of the waterfall, while the juvenile recruits below the waterfall have been systematically eaten by the bass.

4.3.3: Biology of *Austroglanis gilli*

This dissertation is one of the first attempts to understand aspects of the ecology of *A. gilli*, of which virtually nothing is currently known (Bills, 1999). *Austroglanis gilli* was found to be a predominantly insectivorous species, with other food items incidental to its diet (Figure 7). The taxa targeted were mostly benthic aquatic insects such as Chironomidae, Baetidae and various larvae of the order Trichoptera (Figure 8). These data support field observations that the species lives and feeds under cobbles and boulders. It also means that the embedded and sandy substrata found in the lower sites, particularly the pools, limit the ability of *A. gilli* to find food (kick sampling was conducted in riffles and not deep pools, so the availability of food there could not be assessed).

Because no age 0+ *A. gilli* were captured in November 2003 or April 2004, it is difficult to make any inferences about the timing or habitat requirements of catfish spawning events in the river. New research to be undertaken by SAIAB and Rhodes University may be able to shed light on this fascinating question in the future.

4.3.4: Biology of *Micropterus dolomieu*

Micropterus dolomieu is a predatory species, which preyed on large aquatic invertebrates such as freshwater crabs and dragonfly larvae (Odonata), as well as large quantities of mayfly nymphs (Ephemeroptera) (Figure 8). Although *M. dolomieu* is well known as a piscivore (de Moor & Bruton, 1988), only one bass stomach was found to contain a fish, and this was probably a cannibalised juvenile bass. There appeared to be a shift in diet according to size, with small bass feeding almost exclusively on baetid nymphs, while large specimens often had crab in their stomachs. Skelton (1993) studied the stomach contents of largemouth bass (*M. salmoides)* in the recently invaded Blindekloof stream in the eastern CFR, and concluded the bass to be opportunistic visual predators, attacking any prey large enough to be seen. This corresponds to the feeding patterns of adult bass in the lower Rondegat River, where the most obvious prey items left in sandy pools devoid of indigenous fish were crabs, odonate larvae and smaller bass.

Large numbers of 0+ juvenile bass (2-4cm) were recorded in the lower river in November 2003, and these appear to have been spawned in early October, although I did not see any bass nests at my study sites during the October survey. *Micropterus dolomieu* deposit eggs in the nest at water temperatures between 16 and 18°C (Scott & Crossman, 1973), and temperatures had already risen above 20°C by October (Appendix 1). It is possible that this annual crop of 0+ bass provides an ephemeral source of food for the larger juveniles and small-adult bass for a short time.

4.4: ASSESSING THE MECHANISM OF BASS IMPACTS

4.4.1: *Micropterus dolomieu* as a predator

Through all but ruling out other potential causes of the current distributions of indigenous fish in the Rondegat, it is now possible to assume that the major and perhaps the only factor responsible for the disappearance of indigenous fish in the lower river is the presence of *M. dolomieu*. Furthermore, the most likely mechanism of this impact is direct predation. *Micropterus dolomieu* is known to alter fish communities through predation in Canadian lakes where its introduction has precipitated the loss of several species from the shore communities (MacRae & Jackson, 2001). Furthermore its relative, the largemouth bass (*M. salmoides*) has been implicated in the loss of South African fish populations through predatory impacts. In the Blindekloof River in the eastern CFR, for instance, *M. salmoides* eradicated indigenous fish from a string of pools, before it was itself removed by nature conservators (Skelton, 1993).

Introduction of similar piscivorous fishes into river systems have resulted in similar decreases in indigenous species around the world. These include the Nile perch (*Lates niloticus*) in Lake Victoria, which may have caused the extinction of 200 species chichlids (Baskin, 1992) and peacock bass (*Cichla ocellaris* Bloch), which is linked to the disturbance of six fish species in Guatamala (Zaret & Paine, 1973).

The recurring feature of these impacts is that alien piscivores, and in the case of this study bass, remove from a system all fish that are small enough to be consumed. The ability of bass to eradicate indigenous fish in South Africa has been a fairly controversial subject in angling circles, especially since bass in the northern parts of the country do not seem to have caused nearly as much damage to native fish communities as they have in the CFR (de Moore & Bruton, 1988). The reason for this is almost certainly linked to the evolutionary history of the CFR ichthyofauna. Having evolved in isolation in a restricted system of rivers, where the most threatening predatory fish is the omnivorous *L. capensis* (Skelton, 1987), CFR species have evolved no behavioural traits to protect them from specialist predatory fish. In contrast, indigenous fish in the northern river systems of the country have evolved in the presence of highly efficient piscivorous fish species such as the tigerfish (*Hydrocynus vittatis* Castelnau) and the sharptooth catfish (*C. gariepinus*) (Skelton, 1987; de Moore & Bruton, 1988). The result of this isolation is clear in the 'naïve' behaviour of the redfins of the Rondegat River, in that they tend to swim slowly in open water and seldom retreat into cover, even when a diver swims right up to them. This pattern of vulnerability in isolated fish populations is also visible in the galaxiids of New Zealand (Townsend & Crowl, 1991) and Australia (Lintermans, 2000).

Since bass have probably been in the lower Rondegat River since the 1950s, it is no surprise that there are today no redfin minnows or rock catfish visible in the majority of these reaches. It also explains why seeking evidence of direct predation on indigenous fish through bass stomach analysis is a futile endeavour, as Skelton (1993) found when examining the diets of bass in the Blindekloof River. Here, even though bass had been in the river for little over a year, they had already ceased to feed on indigenous fish, as there were no longer any left to prey upon (Skelton, 1993). The best place to find bass still feeding on indigenous fish in the Rondegat River was the pool directly below the waterfall barrier, approximately 100m upstream of the Rooidraai riffle and pool sites (Figure 1). Six bass were caught in this pool over the course of fieldwork, and none of them were found to have fish in their stomachs, though this was probably just an indication that fish had not recently spilled over the waterfall.

The rarity of yellowfish juvenile recruits in the lower Rondegat suggests that the *L. capensis* population there now depends solely on immigration from above the waterfall for recruitment. How, though, one might ask, is it possible for *M. dolomieu* to so efficiently decimate the annual crop of age 0+ *L. capensis* spawned in the lower river, given the relatively low density of bass for much of the year? The key may lie in the coordination of the two species' spawning events.

Females of *M. dolomieu* generally deposit eggs in the nest as soon as water temperatures reach 16°C (Scott & Crossman, 1973), which in the Rondegat River was between the September and October field trips (Appendix 1). The 0+ bass seen in the river in November 2003 were all between 2 and 5cm in length, whereas the average length of cyprinid larvae present in the river at that time was 2cm. There may therefore be a sequence of predation events that ultimately destroy annual yellowfish recruits. When the 0+ yellowfish emerge as free embryos (October-November), the river already contains many of post-larval and juvenile bass. The growth rate of *M. dolomieu* after hatching is dependent largely on discharge and temperature and in favourable conditions, 0+ bass can grow to 8cm in two months (Sabo & Orth, 1995). By the time they reach 1.5cm, age 0+ *M. dolomieu* have developed enough that they can feed on prey as wide as, and in some cases wider, than the gape of their mouths (Easton & Orth, 1992). This means that the bass juveniles would place a substantial predatory pressure on the emerging yellowfish larvae. This phenomenon has been suggested as occurring in New Zealand where comparatively larger alien invasive 0+ trout are thought to prey on smaller native galaxiid larvae as they emerge from the substratum (McDowall, 2003). In the CFR, Harrison (1962/1963) attributed the rapid growth of 4-5cm *M. dolomieu* juveniles in the Berg River to their having "done well on small *Barbus* fry". Even though the pressure of predation from 0+ bass may decrease as they themselves are preyed upon by adult bass, it would be only a matter of time before the yellowfish larvae become large enough to be noticeable to the adult bass, which would then proceed to deplete their numbers further.

4.4.2: Potential factors exacerbating the predatory impacts of *Micropterus dolomieu*

Indications from other rivers in the CFR suggest that small cyprinids, due to their inability to utilise cover or behave defensively, are always vulnerable to invasive predators regardless of the habitat conditions of the river. For example, the Witte River in Bainskloof is a pristine mountain stream, with no agricultural or biological impacts except the presence of *M. dolomieu*, and yet in the invaded reaches indigenous fish have all but disappeared (Shelton, 2003). However, it is widely accepted that many species are threatened not just by alien fish but by a 'cocktail of threats' that includes habitat degradation and habitat loss. At the 2004 South African Society of Aquatic Scientists conference, the Berg-Breede whitefish (*Barbus andrewi*) was declared extinct in the Berg River system, a result of the combined pressures of alien fish, pollution, eutrophication and water abstraction (Buthelezi & Impson, 2004).

Aspects of water chemistry in the Rondegat were not specifically measured during fieldwork, but conductivity, which can be used as a proxy measure for the salt concentration of the river water, was taken. Conductivity within the Cedarberg Wilderness Area never exceeded $60\mu S.cm^{-1}$, whereas in the farmlands of Grootkloof and Keurbos, the conductivity could reach $160\mu S.cm^{-1}$ (Appendix 1). Both these values are regarded as low (Cambray *et al.*, 1997) and the downstream increase in salts can be attributed to changes in geology and evaporative concentration.

The high densities of filamentous algae in the river at sites 4 and 5 in April (Table 17), were probably also the result of summer low flows, together with increased nutrient levels from agricultural runoff and cattle faeces, as well as an abundance of cobble-boulder substrata to grow on. It is unclear whether such changes in water chemistry would have had any effect on the fish in the river.

In the lower river, habitat analysis revealed sedimentation to be a major feature of pools, and to a lesser extent of riffles. The benthos is dominated by sand deposits, which for example at the Site 7 pool are deeper than a meter in places. Similarly, the riffles in the lower river, while not containing large sand deposits, have a highly embedded cobble substratum. What this means is that these sites have minimal benthic cover for *A. gilli*, a species that may rely heavily on such cover to avoid bass predation.

Benthic cover has been found to be a crucial factor controlling the impact of the alien invasive Sacramento pikeminnow (*Ptychocheilus grandis* Ayers) on indigenous benthic sculpins (*Cottus* spp.) in the Eel River of California. Predation experiments showed that tethered sculpins could avoid predation by pikeminnows by hiding under artificial benthic cover structures, which were created as a representation of complex cobble substrata (White & Harvey, 2001). Bills (1999) found that on the Jan Dissels River north of the Rondegat River, *A. gilli* coexisted with *M. dolomieu* for 10km of the lower river. On a similar length of the Heks River, *M. dolomieu* co-existed with *A. gilli* and spotted rock catfish (*A. barnardi*). The conclusion of the study was that sufficient benthic cover from complex rocky substrata allowed the catfish to escape detection by bass during the day (Bills, 1999).

Perhaps the most critical problem with the bass invasion of the Rondegat River however, is that there are no obvious environmental factors that might mitigate against the predatory impacts of bass.

4.4.3: Other potential impacts of bass

Considering other known effects of alien invasive fishes on indigenous fishes (reviewed in Chapter 1, Section 1.1.2), *M. dolomieu* may have affected indigenous fish through disease transfer and competition, as well as by direct predation.

The likelihood of disease transfer in this case is very low. Whereas alien cyprinids such as European carp and Chinese grass carp have been blamed for the introduction of alien parasites to indigenous cyprinid populations (Bruton & Van As, 1986), such afflictions do not generally cross the family barrier. There is for example a monogenean gill parasite that has been introduced into Britain with the largemouth bass, *M. salmoides*. This parasite is, however, like most mongeneans, a genus-specific parasite (Maitland & Price, 1969). There is also a bass virus carried by *M. salmoides* and in some cases other centrarchid fishes, in the USA (Grizzle & Brunner, 2003). This virus as yet has not been recorded in South Africa.

Competition is a plausible impact of the bass invasion, though it is likely that the impacts of direct predation have far outweighed competitive interactions. Today, only large adult *L. capensis* in the lower Rondegat might compete with bass, and due to their divergent diets, and the low densities of adult bass in the river, it is probable that competition between adult bass and yellowfish plays a minor role in structuring the current fish community in the invaded reaches of the river. Ross (1991) made the point that fish species may interact differently with each other at different life history stages, and in this regard it is possible that competition does play a part in the interactions between juvenile bass and yellowfish recruits. This has been found to occur in Italy, where competition between and European pike (*Esox lucius* Linnaeus) and introduced largemouth bass *M. salmoides* appears to have lead to increased mortality in juvenile pike (Lorenzoni et al., 2002). If, therefore, a scenario arose in the timing of spawning events that yellowfish larvae emerged early enough to avoid predation by age 0+ bass, they might still be affected by them through competition for food. All small juvenile bass and yellowfish caught for stomach analyses were found to prey almost exclusively on baetid and chironomid larvae. Although these taxa were relatively common in the lower river (Tables 11 & 12), the possibility of competitive interactions cannot be discounted.

4.5: ALIEN TREES AND THE IMPORTANCE OF THE RIPARIAN ZONE TO FISH IN THE RONDEGAT RIVER

When this project was originally proposed, an aim of the research was to also investigate the effects, whether direct or indirect, of alien invasive trees on the indigenous fishes. What has emerged from the study is that alien tree infestations appear to have had little if any effect on abundances of indigenous fish. In fact, counts of *A. gilli* were consistently higher at the alien tree-infested Sites 3 and 4 than they were in the upper sites where the riparian zone was characterised by pristine fynbos (Table 5). It is difficult to find an explanation for this distribution pattern, as the species' abundance did not correlate with any measured habitat variables. It may in fact be a function of river size, as the riffles and runs at Sites 3 and 4 were deeper and broader than those sampled at the upper sites in the CWA.

The infestation of *A. mearnsii* may actually be having a positive effect on species such as *P. phlegethon* by increasing the amount of submerged woody cover. Woody debris is considered a critical habitat for riverine fish in that it provides hydraulic cover and anti-predatory cover, as well as visual isolation from other fish, which may decrease competition (Crook & Robertson, 1999). *Pseudobarbus phlegethon* was correlated positively with both woody and fine instream structural cover (Tables 7 & 8), and in pool surveys was often seen in close proximity to submerged woody debris. This is corroborated by the microhabitat surveys of Cambray *et al.* (1997) who showed *P. phlegethon* to have a definite preference for submerged structural cover. In this way, it may be that the infestations of *A. mearnsii* are beneficial to *P. phlegethon*, though this would need further investigation.

A potential negative impact of *A. mearnsii* highlighted by Manicom (1999) is that infestations may influence the composition of invertebrates found in the river. Manicom's study suggested, in comparing non-infested to infested reaches of the upper Rondegat River, that certain Ephemeroptera were more common in the alien-infested zone due to increase food availability, while other taxa, such as Notonemouridae and Simuliidae, were less common due to a reduction in habitat quality. The results of benthic invertebrate sampling in the present study showed more taxa to be present in the benthos at the alien-infested Site 3 than in the fynbos-dominated Site 1 (Table 13). In terms of real invertebrate biodiversity effects this result is inconclusive because of the coarse nature of the sampling (semi-quantitative, identification to family only) and it is hoped that the TMF Rondegat River invertebrate project will shed more light on this complex question. The broad finding of the present study, however, remains that alien trees did not have a noticeable negative impact on the river's aquatic animal community.

The river banks at *A. mearnsii*-infested sites were characterised by undercut banks, indicating that some erosion of the bank had occurred. *A. mearnsii* is well known to cause erosion and sedimentation because of its shallow root structure and tendency to crowd out other bank-stabilising riparian plants (Ractliffe *et al*, 2003), and may in part be responsible for the increasing sedimentation found at sites downstream.

Acacia mearnsii is not by any means the only contributor to sedimentation in the lower Rondegat River, however. At the Site 7 pool on Keurbos Farm, constant trampling by cattle has reduced the north bank of the pool to a crumbling sandbank.

Although the grass sown in the pastures to replace *A. mearnsii* is considered a good trapper of sediment (February, 2002) in areas where the cattle wander frequently the grass is unable to fulfil this function. The cattle are also likely to be damaging the marginal vegetation of the river and altering instream habitat through bank widening (Belsky *et al.*, 1999).

There is also another potential reason for the substantial amount of sand recorded at the Keurbos Farm sites. In an interview given in 2002, the farmer at Keurbos, Mr. J.H. Nieuwoudt, claimed that the fires started both by accident and intentionally by nature conservators in the CWA in recent years had been followed by massive loads of sand coming down the river when it flooded, and being deposited in the pools close to his house (February, 2002). It seems that the project currently underway to remove the plantations of pines above Algeria, coupled with fynbos fires, may have resulted in heavy erosion of the upper catchment, a phenomenon that has been recorded elsewhere. For example, a wildfire in the upper catchment of the Malibu Creek system in California, resulted in substantial fine sediment loads being deposited lower down in the river in later winter floods (Spina & Tormey, 2000). The sediment deposited seasonally in the upper catchment of the Rondegat River appears to have spread all the way down the river, accumulating in the large pools at Rooidraai and Keurbos and in the small canyon upstream of the Clanwilliam reservoir, adding to the sediments already deposited there as a result of erosion from bank degradation.

It therefore seems difficult attribute the sediment levels present in the lower river to any one thing. It is clear, however, that for the quality of indigenous fish habitat to improve, the riparian zone of the entire river, not just the alien-tree-infested sections of it, will need to be critically re-evaluated. In his assessment of the environmental state of the catchment, February (2002) concluded that a riparian buffer zone of 30m should be maintained between the river channel and any agricultural activities. While orchards in the catchment are for the most part an acceptable distance from the river, the gaps between them are generally either filled with alien invasive plants, or with pastures containing cattle that have reduced the effective buffer zone to one or two meters of mixed alien and indigenous riparian trees.

4.6: CONCLUSIONS

This study has succeeded for the first time in critically and quantitatively demonstrating the severe impact smallmouth bass (*M. dolomieu*) can have on the indigenous fish assemblage of a CFR mountain stream. If one reviews the null hypotheses set out at the beginning of the project (see Chapter 1.4), one can now state the following:

H_0A: Measured aspects of physical habitat do not differ between bass-invaded and non-invaded sites. This null hypothesis must be rejected, as bass-invaded sites were found to have higher levels of sediment, a degradation of physical habitat quality with the potential to influence fish distributions.

H_0B: If some aspects of physical habitat do differ between bass-invaded and non-invaded sites, then none of the indigenous fish species are limited in their distributions by these factors. This null hypothesis cannot be rejected, as linear regressions at non-invaded sites suggested that none of the indigenous species are affected by sedimentation alone. Sedimentation may, however, have exacerbated the predatory impacts of *M. dolomieu* on *A. gilli* in the lower river.

H_0C: No species of indigenous fish show a preference for a single limiting food type. This null hypothesis cannot be rejected, as all fish species were found to be opportunistic feeders.

H_0D: If some species of indigenous fish do prefer certain food types, the availability of these food types does not differ between bass-invaded sites and non-invaded sites. This null hypothesis cannot be rejected, as aquatic invertebrates, the main food source of all fish species, were found in relatively equal densities throughout the river.

Thus, it is possible to attribute the absence of indigenous fish in the lower Rondegat River primarily to the presence of the bass. In the case of *A. gilli*, the loss of benthic cover by increased sedimentation appears to have acted in synergy with bass predation to exclude the species from the lower river, since evidence from other Cedarberg rivers suggests that the species can co-exist where there is sufficient benthic cover for *A. gilli* (Bills, 1999). The only fish presently capable of co-existing with *M. dolomieu* in the lower Rondegat River are adult *L. capensis*. This species is however also potentially in jeopardy, since *M. dolomieu* appear to be destroying the annual recruitment of juveniles in infested reaches of the river, with the result that the population of *L. capensis* in these reaches is able to survive only through immigration of adults from upstream of the bass barrier. This study reaffirms the perception that physical barriers such as the waterfall on the Rondegat River are currently the only effective mechanisms preventing alien predators like *M. dolomieu* from eradicating indigenous fish species throughout CFR river systems.

During the course of sampling and data analysis, some hard lessons were learned regarding the difficulty in effectively estimating fish populations. Although the literature is full of studies that sought to maximise the accuracy of fish counts, no research has been published that tests these theories in a South African context. For example, the low conductivity and difficult geomorphology of the Rondegat River made it practically impossible to successfully use most of the electrofishing techniques prescribed by the literature. Ultimately, a method was invented for this study that was only accurate enough to detect relative abundances between riffle sites. Research into an optimum method of electrofishing CFR streams that is environmentally friendly is critically needed. Another problem encountered during data analysis was that since neither electrofishing nor snorkelling could be properly tested for their accuracy, the data collected from riffles and pools could not be compared. While these problems did not affect the significance of the final result, they offer a warning to other researchers, particularly those that seek to detect habitat preferences in fish. In order to do so, sufficient understanding of the scale and intensity of sampling required is critical, as is the need to properly calibrate the accuracy of one's fish sampling technique in all habitats before embarking on data collection.

Another significant finding of this project is that while habitat quality may be important for structuring local fish communities, infestations of alien black wattle (*A. mearnsii*) in the riparian zone do not appear to have a significant impact on habitat quality when compared to other environmental disturbances in the catchment, such as cattle farming. Ironically, it appears that the clearing of alien pine plantations, coupled with frequent fynbos fires in the upper catchment, may have caused more damage to the instream habitat of the Rondegat River through sedimentation than the living infestations of *A. mearnsii*. This is an important consideration for managers, in that the effects of alien tree clearing on catchment erosion should be seriously considered when embarking on such operations, and mitigating measures be put in place.

Ultimately this study has shown that *M. dolomieu* is a critical threat to indigenous fish in the CFR and that this threat is likely to be a function of the evolved vulnerability of these species to alien predators. It is, however, important to keep in mind that other alien invasive fish species, which do not prey directly on indigenous species, may also have a negative influence on the indigenous fish of this region, especially if combined with other factors such as habitat degradation. When investigating the serious situation that the indigenous species of the CFR face, it is crucial that all aspects affecting their survival are investigated, if a solution is to be found.

CHAPTER 5: IMPLICATIONS FOR FUTURE REHABILITATION EFFORTS IN THE RONDEGAT RIVER

5.1: CHOOSING A REALISTIC REHABILITATION STRATEGY

5.1.1: The eradication of smallmouth bass from the lower river

When embarking on a rehabilitation effort of any kind, it is crucial that the strategy chosen for implementation is reasonable in terms of the funding available, and that it can provide long-term viability. One must therefore always commence rehabilitation with clear goals that will be both logistically achievable and result in a permanent benefit to the environment.

In the case of the Rondegat River fish fauna, the findings of this thesis make it abundantly clear that the eradication of smallmouth bass from the river is by far the most beneficial rehabilitative action one can take. It is however far easier said than done. While no full-scale attempt at eradicating an alien fish species from a reach of river has ever been attempted in South Africa, fish control has long been practiced in North America, both for conservation and to protect and enhance commercial and recreational sport fisheries. There have consequently been many lessons learned, some of which would need to be heeded carefully before an operation were put in motion here.

A review of fish control projects in the United States of America by Meronek *et al.* (1996) found that of 250 projects, only 43% were successful. However, in projects where the goal had been the complete eradication of a species, 63% were successful, whereas where the goal was only to reduce stocks, only 40% of the projects worked (Meronek *et al.*, 1996). From this it is clear that while stock reduction is a generally futile endeavour from a conservation perspective, there is no guarantee that total eradication will be achievable either.

The Rondegat River was highlighted by Bills (1999) as a river with rehabilitation potential, because it seemed an ideal candidate for treatment with piscicides. The reason for this is that it is a relatively short section of river to be treated; the river was for the most part a single channel, and because the invaded segment was generally easy to reach. Other factors making it desirable were its relatively high fish diversity, and the presence of a unique genetic population of *A. gilli* (Impson et *al*., 2000).

Although many chemicals have been used over the years to control fish, the most popular products by far have been Rotenone and Antimycin A. Rotenone is a naturally occurring toxic compound found in leguminous plants (American Fisheries Society, 2000), while Antimycin is a toxin produced by *Streptomyces* bacteria (Finlayson *et al*., 2002). Both chemicals are effective piscicides, which disrupt respiration in a fish's cells and are rapidly absorbed through the gills. They are also biodegradable, their toxic half-life decreasing with rising water temperature and pH (Finlayson *et al*., 2002). Both piscicides have been used with varied success in America and elsewhere.

In Australia, Rotenone was used to great effect to help save the threatened galaxiid *Galaxias olidus*. A section of a mountain stream, separated upstream and downstream from the rest of the river by physical barriers, was treated with rotenone to eradicate the introduced rainbow trout that had been stocked in the river. A single treatment managed to eradicate the trout, and in the years that followed, *G. olidus* re-colonised the reach left vacant by the trout (Lintermans, 2000). In Arnica Creek, a tributary of Yellowstone Lake, USA, Antimycin was successfully used to eradicate brook trout in order to free the river for recolonisation by the native cutthroat trout (Gresswell, 1991).

With regards to the choice of toxin, Animycin, while more expensive than Rotenone, is extremely toxic to fish (Finlayson *et al*., 2002) This means it can be used at extremely low concentrations, which has the added benefit of causing less damage to non-target organisms such as invertebrates. Another bonus of Antimycin is that, unlike Rotenone, it does not repel fish, making the application of the piscicide to localised areas far more effective (Finlayson *et al*., 2002). A potential drawback to Antimycin is that it breaks down more rapidly than Rotenone, and requires more input stations in the field (R. Bills, pers. com. 2004). A critical lesson learned from projects like that at Arnica Creek, is that a fish eradication requires careful planning and pre-treatment toxicity testing to ensure effectiveness; a large team of highly trained individuals, co-ordinated to ensure correct application, and effective follow-up to ensure the project's success (Gresswell, 1991).

Even if bass were to be successfully removed from the Rondegat River, the key would be to prevent re-invasion. When this project was first proposed, following the recommendations of Impson *et al.* (2000), a key feature was the construction of a barrier weir near the high-water mark of the Clanwilliam dam, on a bedrock shelf approximately 500m upstream of the road bridge (Figure 2), to ensure the maximum reclamation of invaded river. However, the proposed raising of the Clanwilliam dam wall by ten meters (D. Impson, CapeNature, pers. com. 2004) now means the originally planned site could be inundated when the reservoir is at 100% capacity, rendering the bass-barrier useless. Furthermore, the lower section of the river that flows through the canyon above the reservoir is already in jeopardy as a result of an abstraction weir placed a further 1km upstream of the proposed weir site (Figure 1). This weir is a contentious structure in the catchment, and has already been the subject of a court case between the owner of the riparian land and the owners of the weir, who farm citrus on the banks of the reservoir (February, 2002). Although currently in a state of disrepair, this weir abstracts so much water from the river in the dry months, that when the river was visited in February 2004, it had run dry downstream of the weir. Ironically, the weir seems also to have aided the conservation of the Rondegat River, by preventing the invasion of bluegill (*L. macrochirus*) which have never been recorded upstream of the weir.

Since it is unlikely that anything short of protracted legal action will precipitate the removal of this weir from the canyon, it seems the river between it and the reservoir will remain a marginal habitat for fish in the years to come. As a result, the most pragmatic action to take would be to repair the existing weir to make it an effective bass barrier, and exclude the 1500-odd meters below from the river's rehabilitation plan. In terms of preventing other forms of re-invasion, it is critical that the land-owners in the catchment be made aware of the damage caused by the bass, and be encouraged to enforce the prevention of re-introductions of bass by either their employees or by visiting guests to their holiday facilities.

5.1.2: The rehabilitation of the Rondegat River's riparian zone

While the successful eradication of smallmouth bass from the lower river would be of significant benefit to the fauna of the Rondegat River, such an operation in isolation would in no way guarantee the return of a pristine fish community as occurred in the centuries before bass were introduced. The high levels of sediment present in the lower river may hamper the recolonization of *A. gilli*, while other species like *B. calidus* may be affected by the loss of suitable spawning habitat. The problem with sedimentation levels in the river highlighted by this study is that they appear to have been caused by several factors. These include erosion of bare ground in the upper catchment that was created by fires and the removal of pine and eucalyptus plantations, trampling of the riparian belt by cattle, and bank destabilisation by infestations of black wattle (*A.mearnsii*) and blackwood (*A. melanoxylon*) in the riparian zone. If the situation were to be improved, all of these causes would need to be addressed.

The issue of erosion in the upper catchment is a highly contentious subject. For many years up until the present, the managers of the Cedarberg Wilderness Area and the farmers in the lower catchment have had poor relations as a result of the two parties differing attitudes towards controlled burning policies. In the past, CWA managers disapproved of the farmers' patch-burning technique, used for centuries by farmers to improve grazing for cattle. Conversely, the apparent inability of the CWA to control fynbos fires (started either accidentally or for fynbos management) in summer, was scorned by the farmers as damaging to the soil and resulting in massive erosion (February, 2002). Sadly, animosity has built up between the parties as a result of poor communication in the past (pers. obs., 2003). However, the recent appointment of an enthusiastic new manager for the CWA has led to new hope that communication can be resumed on this controversial subject, and that a solution that puts the river's integrity high on its agenda might yet be reached.

The issues of alien riparian trees and cattle in the riparian zone are also complex, as the two threats are intricately intertwined. While government initiatives such as Landcare and Working for Water currently have no clearing operations in the catchment due to financial constraints (pers. obs. 2003; D. Impson, pers. com. 2003), the farmers in the catchment have cleared alien trees on their own lands in order to make room for more pastures in the riparian zone. While this has been of some benefit in that indigenous riparian trees have recolonised the riverbank in places, a large proportion of the riparian zone is now dominated by grass pastures and opportunistic alien species like bramble (*Rubus* sp.). Although the grass that is sown to replace the alien thicket is able to bind the soil and prevent erosion (February, 2002), the trampling of the riverbank by cattle is so severe in some areas as to render the grass useless in this function. Cattle have long been considered a threat to fish habitat in America because they heighten erosion, increase nutrient levels and bacteria through faecal contamination, and ultimately cause channel alteration through bank destabilisation (Belsky *et al*., 1999). These effects were particularly obvious at the study sites on Keurbos Farm, where the river channel itself is a major route of movement for the cattle.

The only certain way properly to manage cattle movements in the riparian zone is by fencing (Platts & Wagstaff, 1984). This allows one to exclude cattle entirely from the degraded areas, and will in time allow the banks and the river to recover. There are, however, several potential problems in taking this approach. The Rondegat valley can become very hot and dry in summer, and although there are some natural springs, wetlands and tributaries scattered along its length, the river is the main source of water for cattle in the dry season.

At the moment, the problem is greatest on Keurbos Farm, where the vast majority of *Acacia mearnsii* has been cleared as part of the farmer's firewood-gathering industry. On the other farms, the dense *A. mearnsii* thickets prevent the cattle from reaching the riparian zone in most places. However, if the *A. mearnsii* were to be properly cleared, the cattle would soon become a problem on those farms as well. When considering the threat posed by alien trees to the river, it is pertinent to mention a finding of February (2002), who interviewed all the farmers in the valley. The perception with many of them was that the alien trees actually improved the flow of water in the river, contrary to recent scientific evidence that shows *A. mearnsii* to cause water loss in the river (Ractliffe *et al*, 2003). This perception could pose a problem, as there might be resistance to a co-ordinated alien-clearing operation being conducted by government projects such as Working for Water or Landcare.

Another problem with fencing is that it may be unattractive to the farmers from a financial standpoint. Fencing is costly to lay and to maintain, and the perceived benefit of doing so, the improvement of habitat for fish, will probably not make much of an incentive. Even the farmer at Keurbos, who is very conservation conscious and is keen to see the bass removed to help the re-establishment of yellowfish and redfins, is likely to have a problem with fencing since it will block access not only for his cattle but may hamper paying guests who might want to utilise the river for recreational fishing. It is critical that for a rehabilitation project to work, the needs of all stakeholders must first be accounted for.

5.1.3: The issue of fish distributions in the upper Rondegat River

When Mr. R. Bills conducted field surveys for his report on the conservation of Austroglanididae in the Olifants-Doring System (Bills, 1999), he surveyed the river from Algeria upstream to a small waterfall, which was the natural upper limit of fish distributions in the Rondegat River. In a pool directly below the waterfall, he recorded *G. zebratus* and *B. calidus*. However, in the reach of river between the waterfall and Algeria, he was not able to find any fish, though the habitat condition seemed good (R. Bills, unpublished data). This led him to believe that the current design of the causeway at the Algeria campsite (Figure 1) made it a significant barrier to fish movement, and might ultimately result in the loss of several kilometres of suitable habitat to fish within the Cedarberg Wilderness Area. This is because fish that may have migrated past Algeria in the past to spawn, can now only pass downstream through the causeway, and cannot return upstream through it (R. Bills, pers. com, 2004).

While the area of the river upstream of Algeria was too remote to include in the sampling strategy for the bulk of this dissertation, an expedition was mounted in September 2004 to confirm the impact of the Algeria causeway. A 10m section of riffle directly below the causeway was cordoned off with stop nets, and was shocked with a DEKA 3000 backpack electrofisher for 5 minutes, using a zig-zagging single pass downstream towards the bottom seine, where flushed fish were captured in the net's bag. The same procedure was followed in a riffle/run segment immediately upstream of the swimming pool, which is formed by the causeway. The comparative survey results of this exercise were striking (Table 18). *Austroglanis gilli* had all but disappeared from the upper segment, while the numbers of redfins were too low to be conclusive of anything. This does however suggest that the causeway is a serious barrier to movement of *A. gilli*, as the habitat above the causeway appeared to be perfectly adequate for the species. Another interesting point was that several ghost frog (*Heliophryne* sp.) tadpoles were caught in the upper segment. Large adult *A. gilli* are thought to prey on this species (Bills, 1999). The tadpoles were absent below the causeway, and were in fact never caught in riffles during the project's seasonal site surveys downstream. This finding suggests that *A. gilli* have been scarce above the causeway for a considerable time.

Table 18: Comparative abundances of fish species captured above and below Algeria campsite causway using single-pass electrofishing with block nets over 10 m. Survey conducted 13 September, 2004.

Species	Below Causeway	Above Causeway
Austroglanis gilli	18	1
Pseudobarbus phlegethon	2	0
Barbus calidus	1	1

From these findings, it would appear that a method for allowing fish to bypass the causeway should be developed, and the best solution would be a fish ladder of some description. This could prove problematic, in that the causeway already has a mechanised sluice in its centre to allow water to pass through (Appendix 3; Plate 11). This is opened partially in summer to allow the swimming pool to form, and fully in winter, in order for the river to return to its natural state. The problem with the current system however, is that there is a broad concrete shelf between the sluice and the river downstream, so that even when the sluice is fully opened, it forms a rapid chute, which is likely to prevent the passage of fish. A fish ladder will have to be designed that does not compromise the sluice mechanism, but at the same time allows fish to pass beneath the surface of the causeway, so that passage of vehicular traffic is not affected.

5.2 MANAGEMENT RECOMMENDATIONS FOR REHABILITATING THE RONDEGAT RIVER

5.2.1: The planning stage

- Before any management plan can be applied to the Rondegat River, it is critical that a conservancy be formed between the landowners in the catchment and the custodians of the Cedarberg Wilderness Area CWA). In order to facilitate the formation of this conservancy, planning meetings should be arranged to address the concerns of all parties to make the venture mutually beneficial. These meetings should include representatives from the CWA, the Bosdorp community, and all the farmers in the catchment. Issues to be discussed should include:
    - The potential benefits of the conservancy to farmers (for example, tax relief in exchange for changed management policies).
    - The current views of farmers towards CWA fire control policies and methods. This was a serious issue identified by February (2002), which must be addressed in order to strengthen trust and co-operation between CWA and the farmers. The CWA may need to re-assess these policies and implement changes in order to minimise catchment erosion and subsequent sedimentation of the river.
    - The obligation of landowners to remove alien invasive vegetation in accordance with government biodiversity legislation and CARA regulations.
    - The latest scientific evidence that alien vegetation decreases stream flow.
    - The obligation of landowners under the Water Act to register water use and work with the local Catchment Manegement Agency (when it is initiated) to ensure equitable and sustainable utilisation of water.
    - The benefits to the river of managing cattle movement in the riparian zone.
    - The importance of removing bass from the lower river and re-establishing indigenous fish there.

5.2.2. Short term operations

(a) Bass eradication

- Prior to an eradication programme being implemented, the following should first be achieved:
    - The abstraction weir 1km upstream of Clanwilliam dam (Figure 1) should be re-assessed for its ability to function as a year-round bass barrier, and appropriately upgraded if necessary by contractors approved by the conservancy.
    - A dedicated piscicide team, which has been trained in the safe and appropriate use of piscicides and application machinery, should be assembled by CapeNature.
- It would be desirable to conduct the treatment in late summer, when flows are lowest. This would also allow the majority of yellowfish present in the lower river to be removed and placed in temporary storage for the duration of operations. This can be achieved by seining the large Site 5 pool where most of the yellowfish accumulate in summer.
- A neutralising station applying potassium-permangenate to neutralise the piscicide should be set up directly below the abstraction weir, to ensure toxins do not reach the Clanwilliam dam, which is an important resource to the local bass-fishing industry.
- Application of toxins should be conducted at least twice to ensure success. If no bass are caught during the second application, a follow-up application will not be necessary. However, a follow-up survey should be conducted after a significant change in flow (e.g., winter flooding), to ensure that bass have not survived in undetected refugia.

(b) Upgrading the causeway weir at Algeria campsite

- A fish ladder or bypass will need to be constructed on or around the Algeria *swemgat* causeway in order to allow the upstream movement of fish in summer. A specialist engineer will need to be brought in to examine the site and recommend the preferred method.

5.2.3. Medium- to long-term operations

a) Rehabilitating the riparian zone in the farmed reaches of the river

- The rehabilitation of the riparian zone in order to improve instream habitat conditions for the fish will be a complex endeavour that will require excellent co-operation between the landowners and CapeNature.
  - Alien trees need to be re-evaluated in terms of the threat they currently pose to the river. Current legislation means that they ultimately will need to be removed, but it is critical that not all the cleared land is simply converted to pastures.
  - Provided that each farmer considers the land he currently has for grazing sufficient, the alien-tree-infested sections should be fenced off and replanted with indigenous riparian species after clearing.
  - It is strongly recommended that government agencies such as Working for Water or Landcare are involved in alien tree operations, as well as monitoring and follow up, and that they use members of the Bosdorp community for the majority of contracting work. The Bosdorp community should also be involved in fencing and replanting operations in the riparian zone.
  - At Keurbos farm, large sections of the river channel are currently a thoroughfare for the cattle. This needs to be altered by the strategic placement of fencing in the riparian zone, which takes into account the current network of cattle trails on the property, and minimises the number of places that cattle can cross the river.

b) Minimising erosion in the CWA

- It is critical, both for the success of a conservancy between the farmers and CapeNature, as well as for the ultimate sustainability of the rehabilitated river, that the current fire control and suppression procedures of all conservancy members be reassessed and, if practical, adjusted. Only then will a serious barrier to co-operation and a potential threat to river ecosystem health be removed. It is also the recommendation of this report that a study be commissioned to assess current erosion levels on CWA land, in order to find new strategies to counteract erosion in the future.

5.3: IMPLICATIONS OF THIS STUDY FOR REHABILITATION EFFORTS IN OTHER CAPE RIVERS

While this chapter has so far looked at the practical steps needed to improve the ecosystem of the Rondegat River, it must be recognised that any rehabilitation projects of this nature must be carried out within the broader geographic and socio-economic context of the catchment. Most stake-holders are likely to see the Rondegat River, as a water-resource first, and a habitat for fish and other organisms second; as water demand increases within the Western Cape, the Rondegat and other rivers are likely to come under more and more pressure. While a broader discussion of the implications of national water use policies and other broad factors that may ultimately affect the continued survival of indigenous fish in the CFR is beyond the scope of this study, the existence of such over-arching influences on the fate of these species must never be forgotten.

In terms on the immediate implications of this study for indigenous fish conservation, my findings suggests that practical interventions such as poisoning alien fish can be both possible and desirable in the Rondegat River, provided that other impacts present in the system are not ignored. Many other rivers in the region suffer the same suite of problems as the Rondegat, but due to differing geography, agricultural practices, or social structures, will require a different approach to each impact to ensure meaningful improvement. In some cases, such as where alien fish co-exist with indigenous fish, methods other than eradication using piscicide will have to be investigated to remove or at least control the alien fish. In terms of the threat posed by alien invasive fishes in other rivers of the Western Cape, it is critical that steps be taken very soon to improve the current situation. The recent extinctions of *B. andrewi* from the Berg River (Buthelezi & Impson, 2004) and of *P. phlegethon* from the Jan Dissels River (Bills, 1999) are two examples that serve to illustrate the seriousness of the situation, and that action is now needed. Logistics will always be a fundamental limiting factor in alien fish removal operations, particularly when dealing with the problem of invasive species within a large catchment. Alien invasive species such as *M. dolomieu* are present in many rivers in the CFR, but very few of these rivers are as accessible as the Rondegat, and many cannot even be reached by off-road vehicles. In this case, careful assessment is needed of each affected river system so that rivers with both conservation-priority status and practical rehabilitation potential can be identified. This work is currently underway through the Alien Fish Project of the Table Mountain Fund, of which this study was a component, and it is hoped that in the years to come, the findings of these studies will result in the reclamation of critical rivers for their unique and threatened fish fauna.

In ongoing endeavours to control the spread of alien fish, both in the Western Cape and in South Africa in general, it is crucial that the recreational angling community is kept involved at all times. Only when anglers are properly informed and do not feel antagonised, are they likely to co-operate with alien fish control programmes. It is therefore important to maintain a steady line of communication with organisations such as the Cape Piscatorial Society, and in particular local bass fishing clubs, so that they can know the reasons behind these operations, and will not just blindly believe that CapeNature is "out to get their bass". They should be reassured that eradication programmes will for the most part not affect established fishing waters (e.g. Clanwilliam dam), but know that they should be responsible anglers and not move alien fish around without a permit. Only through their co-operation will eradication programmes ultimately succeed.

The removal of alien fish will seldom be the only step required to save a fish population. Habitat preservation is a non-negotiable factor in fish conservation, and the key to any holistic river rehabilitation strategy will be to gain the support of all affected role-players, and most critically the landowners, in order to achieve meaningful results. It is pointless to dictate ecological principles to a farmer whose chief concern is to keep his farm solvent. Government support in the form of tax relief in exchange for conservation practices is an important tool in this situation and can show serious benefits, as has been demonstrated by CapeNature's Stewardship Programme for the co-operative conservation of renosterveld, which has seen many acres of highly threatened vegetation safeguarded through the incentivised co-operation of the land-owner (CNC/CEPF/NBI, 2003).

Only through innovative thinking, dedication, appropriate funding and co-operation will the unique and highly threatened fish fauna of the Cape Floristic Region be preserved, so that future generations can enjoy the continued existence of this fundamental part of our natural heritage.

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APPENDICES

Appendix 1: Physico-chemical data

a) September 2003

Site Measurements

	Conductivity ($\mu S.cm^{-1}$)	Temperature (°C)	pH
Algeria Riffle	24.8	14	5.5
Rockface Riffle	21.4	12.6	5.8
Meadow Riffle	31.5	14.5	5.7
Cottage Riffle	33.2	15.2	5.5
Rooidraai Riffle	35.5	15.2	5.9
Upper Keurbos Riffle	37.3	15.5	5.7
Lower Keubos Riffle	39	16	5.8
Canyon Riffle	34.7	13.6	6.1

Maximum/minimum temperatures

	Max	Min
Site 1: Algeria	15	10
Site 5: Rooidraai Pool	15	12
Site 7: Keurbos	15	14
Site 8: Canyon	no data recorded	

b) October 2003

Site Measurements

	Conductivity ($\mu S.cm^{-1}$)	Temperature (°C)	pH
Algeria Riffle	20.5	16.3	5.5
Rockface Riffle	21.6	16	5.8
Meadow Riffle	32.5	17.5	5.8
Cottage Riffle	33	16.5	6.0
Rooidraai Riffle	37	20.6	6.1
Upper Keurbos Riffle	33.7	16.8	61
Lower Keubos Riffle	38.9	22	6.1
Canyon Riffle	44.4	18	6.3

Maximum/minimum temperatures

	Max	Min
Site 1: Algeria	21	12
Site 5: Rooidraai Pool	21	14
Site 7: Keurbos	22	14
Site 8: Canyon	23	14

c) November 2003

Site Measurements

	Conductivity ($\mu S.cm^{-1}$)	Temperature (°C)	pH
Algeria Riffle	46.9	21.2	5.6
Rockface Riffle	48	19.6	5.7
Meadow Riffle	102.4	22.3	5.9
Cottage Riffle	103	24	5.6
Rooidraai Riffle	105.8	24	5.4
Upper Keurbos Riffle	118.7	28	6.0
Lower Keubos Riffle	120.4	26.3	5.6
Canyon Riffle	130.3	23.9	5.9

Maximum/minimum temperatures

	Max	Min
Site 1: Algeria	26	16
Site 5: Rooidraai Pool	24	19.5
Site 7: Keurbos	equipment failure	
Site 8: Canyon	26	20

d) April 2004

Site Measurements

	Conductivity ($\mu S.cm^{-1}$)	Temperature (°C)	pH
Algeria Riffle	51	14.4	5.8
Rockface Riffle	52.7	17.7	5.72
Meadow Riffle	102	18.4	5.84
Cottage Riffle	103	14.6	5.86
Rooidraai Riffle	111	16.7	6.22
Upper Keurbos Riffle	118.8	17.5	5.96
Lower Keubos Riffle	118.8	17.5	5.96
Canyon Riffle	147.7	17.5	6.22

Maximum/minimum temperatures

	Max	Min
Site 1: Algeria	22.5	9.5
Site 5: Rooidraai Pool	17.5	14.5
Site 7: Keurbos	20	15
Site 8: Canyon	no data recorded	

Appendix 2: Fish gut contents

a) *Labeobarbus capensis*

Code	RABLC1	Date	17-10-03
Species	*L. capensis*	Site	1
Length (cm TL)	18.5	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Fish caught in drift net		
Taxon	Number	Taxon	Number
Corixidae	1		
Baetidae	5		
Elmidae	3		

Code	RABLC2	Date	30-04-04
Species	*L. capensis*	Site	1
Length (cm TL)	16.5	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	8	Detritis	92
Algae		Plant material	
Notes	Fish caught in drift net		
Taxon	Number	Taxon	Number
Baetidae	2		
Terr. Coleoptera	1		
Corixidae	1		

Code	RABLC3	Date	30-04-04
Species	*L. capensis*	Site	1
Length (cm TL)	15.6	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	60	Detritis	40
Algae		Plant material	
Notes	Fish caught in drift net		
Taxon	Number	Taxon	Number
Baetidae	10		
Simuliidae	1		
Gryrinidae	1		

Code	RABLC5	Date	30-04-04
Species	*L. capensis*	Site	1
Length (cm TL)	8.3	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	26	Detritis	
Algae	74	Plant material	
Notes			
Taxon	Number	Taxon	Number
Corixidae	1		
Chironomidae	4		
Baetidae	4		
Elmidae	1		
Trichoptera	1		
Terr. Coleoptera	1		

Code	RMWLC5	Date	20-10-03
Species	*L. capensis*	Site	3
Length (cm TL)	6.5	Weight (g)	2
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	30		
Ecnomidae	1		
Ephemeroptera	1		
Trichoptera	1		

Code	RMWLC6	Date	20-10-03
Species	*L. capensis*	Site	3
Length (cm TL)	11.9	Weight (g)	12
Food Type	% Volume	Food Type	% Volume
Invertebrates	97	Detritis	3
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	17		
Chironomidae	80		
Baetidae	10		

Code	RMWLC7	Date	20-10-03
Species	*L. capensis*	Site	3
Length (cm TL)	11.4	Weight (g)	10
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	10		
Trichoptera	1		
Chironomidae	10		

Code	RMWLC9	Date	20-10-03
Species	*L. capensis*	Site	3
Length (cm TL)	8.6	Weight (g)	6
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	21		
Simuliidae	1		
Baetidae	2		

Code	RMWLC10	Date	24-04-04
Species	*L. capensis*	Site	3
Length (cm TL)	8.7	Weight (g)	5.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Baetidae	2		
Chironomidae	6		
Trichoptera	1		
Simuliidae	1		

Code	RMWLC11	Date	24-04-04
Species	*L. capensis*	Site	3
Length (cm TL)	6	Weight (g)	2.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	3		
Chironomidae	6		

Code	DCTLC3	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	8.6	Weight (g)	5.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	99	Detritis	
Algae		Plant material	1
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Baetidae	25		

Code	DCTLC4	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	9	Weight (g)	5.9
Food Type	% Volume	Food Type	% Volume
Invertebrates	38	Detritis	47
Algae	14	Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Baetidae	4		

Code	DCTLC5	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	8.5	Weight (g)	5.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Libellulidae	2		
Corixidae	6		
Baetidae	4		

Code	DCTLC6	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	11	Weight (g)	10.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	13	Detritis	87
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Ephemeroptera	5		
Trichoptera	1		
Chironomidae	3		

Code	DCTLC7	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	9.2	Weight (g)	6.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	91	Plant material	9
Algae			
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	10		

Code	DCTLC8	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	15	Weight (g)	27.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	5	Detritis	82
Algae	13	Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Simuliidae	4		
Chironomidae	1		
Baetidae	2		

Code	DCTLC10	Date	2/4/2004
Species	*L. capensis*	Site	4
Length (cm TL)	14.5	Weight (g)	24.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	2	Detritis	58
Algae	40	Plant material	
Notes			
Taxon	Number	Taxon	Number
Coleoptera	1		

Code	ACTLC1	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	35.5	Weight (g)	366.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	73	Detritis	
Algae		Plant material	27
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Corixidae			200
Ephemeroptera			1
Leptoceridae			5
Chironomidae			1

Code	ACTLC2	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	17.5	Weight (g)	41.7
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	58
Algae		Plant material	42
Notes			
Taxon	Number	Taxon	Number

Code	ACTLC3	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	21.1	Weight (g)	62.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	56	Detritis	44
Algae		Plant material	
Notes	Nematode parasites / sand present		
Taxon	Number	Taxon	Number
Chironomidae	50		
Trichotera	1		
Coleoptera	1		
Ephemeroptera	1		

Code	ACTLC4	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	8	Weight (g)	3.8
Food Type	% Volume	Food Type	% Volume
Invertebrates	29	Detritis	71
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	2		
Simuliidae	1		
Chironomidae	1		

Code	ACTLC5	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	7.2	Weight (g)	2.8
Food Type	% Volume	Food Type	% Volume
Invertebrates	39	Detritis	61
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	4		
Chironomidae	1		

Code	ACTLC6	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	7.7	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	85	Detritis	15
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	5		
Ephemeroptera	1		

Code	ACTLC7	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	7.5	Weight (g)	3.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	85	Detritis	15
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		
Simuliidae	1		

Code	ACTLC8	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	7.1	Weight (g)	2.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	70	Detritis	30
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	1		

Code	ACTLC9	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	11	Weight (g)	9.7
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	100
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Baetidae	1		

Code	ACTLC10	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	11.8	Weight (g)	10.9
Food Type	% Volume	Food Type	% Volume
Invertebrates	68	Detritis	24
Algae	8	Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Terr. Invert	1		
Odonata	1		

Code	ACTLC11	Date	28-11-03
Species	*L. capensis*	Site	4
Length (cm TL)	7	Weight (g)	3.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	60	Detritis	40
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	1		
Chironomidae	1		

Code	ARDLC1	Date	23-10-03
Species	*L. capensis*	Site	5
Length (cm TL)	17.3	Weight (g)	43
Food Type	% Volume	Food Type	% Volume
Invertebrates	78	Detritis	11
Algae		Plant material	11
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	115		
Simuliidae	40		
Cladocera	1		

Code	ARDLC4	Date	30-04-04
Species	*L. capensis*	Site	5
Length (cm TL)	41.5	Weight (g)	640
Food Type	% Volume	Food Type	% Volume
Invertebrates	14	Detritis	77
Algae	1	Plant material	8
Notes			
Taxon	Number	Taxon	Number
Baetidae	5		
Corixidae	9		
Chironomidae	1		
Coleoptera	1		

b) *Pseudobarbus phlegethon*

Code	RABPP1	Date	28-04-04
Species	*P.phlegethon*	Site	1
Length (cm TL)	7.1	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	83	Detritis	16
Algae		Plant material	
Notes	caught in drift net		
Taxon	Number	Taxon	Number
Trichoptera	2		
Simuliidae	1		
Chironomidae	1		

Code	RABPP2	Date	28-04-04
Species	*P.phlegethon*	Site	1
Length (cm TL)	5.8	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Balance malfunction		
Taxon	Number	Taxon	Number
Elmidae	1		
Chironomidae	40		

Code	RMWPP1	Date	20-10-03
Species	*P.phlegethon*	Site	3
Length (cm TL)	5.8	Weight (g)	1
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	sand present		
Taxon	Number	Taxon	Number
Diptera larva	3		
Simuliidae	1		
Chironomidae	50		

Code	RMWPP2	Date	20-10-03
Species	*P.phlegethon*	Site	3
Length (cm TL)	5.1	Weight (g)	1
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	10		
Simuliidae	1		
Ephemeroptera	1		

Code	RMWPP3	Date	20-10-03
Species	*P.phlegethon*	Site	3
Length (cm TL)	5.3	Weight (g)	1
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	3		
Ephemeroptera	1		

Code	RMWPP4	Date	20-10-03
Species	*P.phlegethon*	Site	3
Length (cm TL)	5.3	Weight (g)	2
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	20		
Ephemeroptera	1		

Code	RMWPP5	Date	20-10-03
Species	*P.phlegethon*	Site	3
Length (cm TL)	7.3	Weight (g)	6
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Diptera Larva	1		
Chironomidae	50		

Code	RMWPP8	Date	24-04-04
Species	*P.phlegethon*	Site	3
Length (cm TL)	6.7	Weight (g)	3.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	3		
Trichoptera	2		

Code	RMWPP9	Date	24-04-04
Species	*P.phlegethon*	Site	3
Length (cm TL)	7.5	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		
Baetidae	1		

Code	RMWPP10	Date	24-04-04
Species	*P.phlegethon*	Site	3
Length (cm TL)	5.9	Weight (g)	2.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	78	Detritis	
Algae	21	Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	2		
Chironomidae	6		

Code	RCTPP1	Date	21-11-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	5.1	Weight (g)	1.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Ecnomidae	1		
Chironomidae	50		
Simuliidae	20		
Baetidae	1		

Code	RCTPP2	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.2	Weight (g)	2
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	10		
Simuliidae	1		
Ephemeroptera	1		

Code	RCTPP3	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.5	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	3		
Ephemeroptera	1		

Code	RCTPP4	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.5	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	4		
Chiron. Pupa	1		

Code	RCTPP5	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	5.9	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		
Simuliidae	2		
Terr. Diptera	1		

Code	RCTPP6	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.5	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	22	Detritis	78
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	30		
Simuliidae	3		
Coleptera	1		

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Code	RCTPP7	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.8	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	10		

Code	RCTPP8	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.2	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	58	Detritis	42
Algae		Plant material	
Notes	Cestode parasite present		
Taxon	Number	Taxon	Number
Chironomidae	4		
Simuliidae	1		

Code	RCTPP10	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.9	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	2		
Simuliidae	1		
Ephemeroptera	1		
Coleoptera	1		

Code	RCTPP11	Date	20-10-03
Species	*P.phlegethon*	Site	4
Length (cm TL)	7.4	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Cestode parasite present		
Taxon	Number	Taxon	Number
Simuliidae	4		
Ephemeroptera	1		
Chironomidae	9		

Code	DCTPP1	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	7.2	Weight (g)	3.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	6	Detritis	94
Algae		Plant material	
Notes	sand present		
Taxon	Number	Taxon	Number
Crustacea	10		

Code	DCTPP2	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	7	Weight (g)	3.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	23	Detritis	77
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Ecnomidae	1		
Trichoptera	7		
Chironomidae	15		

Code	DCTPP3	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	7	Weight (g)	3.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	10	Detritis	87
Algae		Plant material	3
Notes			
Taxon	Number	Taxon	Number
Chironomidae	17		

Code	DCTPP4	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.8	Weight (g)	3.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	3	Detritis	97
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	5		
Trichoptera	3		

Code	DCTPP5	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6	Weight (g)	2.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	9	Detritis	91
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	5		

Code	DCTPP6	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	7.5	Weight (g)	3.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	90	Detritis	10
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	5		

Code	DCTPP7	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6	Weight (g)	2.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	30	Detritis	70
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Trichoptera	8		
Chironomidae	3		

Code	DCTPP8	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6	Weight (g)	2.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	24	Detritis	76
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		
Chironomidae	1		

Code	DCTPP9	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.3	Weight (g)	2.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	22	Detritis	78
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	30		

Code	DCTPP10	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.9	Weight (g)	2.9
Food Type	% Volume	Food Type	% Volume
Invertebrates	8	Detritis	76
Algae	16	Plant material	
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	1		

Code	DCTPP11	Date	2/4/2004
Species	*P.phlegethon*	Site	4
Length (cm TL)	6.6	Weight (g)	3
Food Type	% Volume	Food Type	% Volume
Invertebrates	41	Detritis	59
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	7		
Chironomidae	16		

c) ***Barbus calidus***

Code	RABBC2	Date	28-04-04
Species	*B. calidus*	Site	1
Length (cm TL)	8.6	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Terr. Invert.			1
Baetidae			9
Elmidae			1
Simuliidae			1

Code	RABBC3	Date	28-04-04
Species	*B. calidus*	Site	1
Length (cm TL)	5.7	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	
Algae	100	Plant material	
Notes	Caught in drift net		
Taxon	Number	Taxon	Number

Code	RABBC4	Date	28-04-04
Species	*B. calidus*	Site	1
Length (cm TL)	6.5	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes	Caught in drift net		
Taxon	Number	Taxon	Number
Simuliidae	1		

Code	RABBC5	Date	28-04-04
Species	*B. calidus*	Site	1
Length (cm TL)	6.6	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes	Caught in drift net		
Taxon	Number	Taxon	Number
Hymenoptera	1		

Code	RABBC6	Date	28-04-04
Species	*B. calidus*	Site	1
Length (cm TL)	6.5	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	29	Detritis	71
Algae		Plant material	
Notes	Drift net		
Taxon	Number	Taxon	Number
Coleoptera	1		
Trichoptera	1		

Code	RMWBC1	Date	24-04-04
Species	*B. calidus*	Site	3
Length (cm TL)	8.5	Weight (g)	5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Coleoptera	1		

Code	RMWBC2	Date	20-10-03
Species	*B. calidus*	Site	3
Length (cm TL)	8	Weight (g)	6
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	1		
Terr. Coleoptera	2		

Code	RMWBC3	Date	20-10-03
Species	*B. calidus*	Site	3
Length (cm TL)	10	Weight (g)	8.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	97	Algae	3
Notes			
Taxon	Number	Taxon	Number
Diptera larva	2		
Baetidae	3		
Heptageniidae	1		
Trichoptera	1		
Terr. Diptera	1		

Code	RMWBC4	Date	20-10-03
Species	*B. calidus*	Site	3
Length (cm TL)	9	Weight (g)	5.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		

Code	RMWBC5	Date	20-10-03
Species	*B. calidus*	Site	3
Length (cm TL)	9.9	Weight (g)	8.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	98	Plant material	2
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	1		
Terr. Invert.	1		

Code	RCTBC1	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	7.7	Weight (g)	3.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	3		
Terr. Coleoptera	5		
Chironomidae	20		

Code	RCTBC2	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	9.1	Weight (g)	6.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	96	Detritis	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Corydalidae	1		
Simuliidae	3		
Terr. Coleoptera	1		

Code	RCTBC3	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	9.8	Weight (g)	6
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Terr. Coleoptera	1		
Terr. Inv	1		
Hymenoptera	10		

Code	RCTBC4	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	7.9	Weight (g)	3.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	95	Detritis	
Algae	5	Plant material	
Notes			
Taxon	Number	Taxon	Number
Terr. Coleoptera	1		
Hymenoptera	1		
Trichoptera	1		
Baetidae	1		

Code	RCTBC5	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	10.3	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Terr. Coleoptera	1		
Hymenoptera	1		

Code	RCTBC6	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	8.2	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Dipteran Larva	1		
Simuliidae	1		
Chironomidae	5		
Trichoptera	1		
Terr. Coleoptera	1		

Code	RCTBC7	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	9	Weight (g)	6
Food Type	% Volume	Food Type	% Volume
Invertebrates	4	Detritis	
Algae	61	Plant material	35
Notes			
Taxon	Number	Taxon	Number
Chironomidae	1		
Trichoptera	1		

Code	RCTBC8	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	8.9	Weight (g)	4.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	24	Detritis	
Algae		Plant material	76
Notes	Unidentified invertebrate fragments		
Taxon	Number	Taxon	Number

Code	RCTBC9	Date	26-04-04
Species	*B. calidus*	Site	4
Length (cm TL)	6.8	Weight (g)	2.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Terr. Invert.	1		

Code	DCTBC3	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	7.2	Weight (g)	3.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	4	Detritis	
Algae	96	Plant material	
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	1		

Code	DCTBC4	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	8	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	
Algae	100	Plant material	
Notes			
Taxon	Number	Taxon	Number

Code	DCTBC5	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	8.4	Weight (g)	5.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	3	Detritis	
Algae	97	Plant material	
Notes	Unidentified fragments		
Taxon	Number	Taxon	Number

Code	DCTBC6	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	8.5	Weight (g)	5.2
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	
Algae	100	Plant material	
Notes			
Taxon	Number	Taxon	Number

Code	DCTBC7	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	8.8	Weight (g)	5.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	8	Detritis	
Algae	92	Plant material	
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	1		

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Code	DCTBC8	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	8.9	Weight (g)	6.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	30	Detritis	6
Algae	64	Plant material	
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	5		
Hymenoptera	1		
Chironomidae	1		

Code	DCTBC9	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	9.7	Weight (g)	7.9
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	
Algae	100	Plant material	
Notes			
Taxon	Number	Taxon	Number
Hymenoptera	2		
Ephemeroptera	1		

Code	DCTBC10	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	9.7	Weight (g)	7.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	2	Detritis	
Algae	98	Plant material	
Notes			
Taxon	Number	Taxon	Number
Hymenoptera	2		

Code	DCTBC11	Date	2/4/2004
Species	*B. calidus*	Site	4
Length (cm TL)	9.7	Weight (g)	7.6
Food Type	% Volume	Food Type	% Volume
Invertebrates	36	Detritis	
Algae	64	Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Hymenoptera	1		
Trichoptera	1		

Code	ACTBC21	Date	28-11-03
Species	*B. calidus*	Site	4
Length (cm TL)	11.4	Weight (g)	10.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Chironomidae	1		

Code	ACTBC24	Date	28-11-03
Species	*B. calidus*	Site	4
Length (cm TL)	11.5	Weight (g)	11.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	1		

Code	ACTBC27	Date	28-11-03
Species	*B. calidus*	Site	4
Length (cm TL)	8.5	Weight (g)	5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Coleoptera	1		

Code	ACTBC4	Date	28-11-03
Species	*B. calidus*	Site	4
Length (cm TL)	9.5	Weight (g)	
Food Type	% Volume	Food Type	% Volume
Invertebrates		Plant material	100
Notes			
Taxon	Number	Taxon	Number

Code	ACTBC5	Date	28-11-03
Species	*B. calidus*	Site	4
Length (cm TL)	9.8	Weight (g)	8.6
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	
Algae		Plant material	100
Notes			

d) *Austroglanis gilli*

Code	RABAG11	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	11.3	Weight (g)	16
Food Type	% Volume	Food Type	% Volume
Invertebrates	93	Detritis	7
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Tricoptera	2	Leptoceridae	5
Baetidae	3	Chironomidae	13
Elmidae	1	Ephemeroptera	1
Teloganodidae	3		

Code	RABAG2	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	7.7	Weight (g)	5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Leptophlebiidae	1	Chironomidae	15
Terr. Coleoptera	1	Chiron. Pupa	1
Elmidae	2		

Code	RABAG3	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	10	Weight (g)	9
Food Type	% Volume	Food Type	% Volume
Invertebrates	89	Detritis	11
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Baetidae	5	Terr. Coleoptera	1
Chironomidae	50	Lepidoptera	1
Trycorythidae	1	Leptoceridae	1
Ephemeroptera	1	Terr. Hemiptera	1

Code	RABAG4	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	4.9	Weight (g)	1
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	1		
Chironomidae	2		

Code	RABAG8	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	11.8	Weight (g)	15
Food Type	% Volume	Food Type	% Volume
Invertebrates	88	Detritis	2
Algae		Plant material	10
Notes			
Taxon	Number	Taxon	Number
Leptoceridae	2	Trichoptera	2
Coleop. Larva	1	Chironomidae	21
Simuliidae	3	Chiron. Pupa	1
Teloganodidae	1		

Code	RABAG9	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	12.5	Weight (g)	19
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Teloganodidae	7	Chironomidae	18
Heptageniidae	1	Baetidae	4
Trichoptera	1	Leptoceridae	2
Simuliidae	7		

Code	RMWAG1	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	13	Weight (g)	26
Food Type	% Volume	Food Type	% Volume
Invertebrates Algae	0.4342	Detritis Plant material	0.4432
Notes	Contents used for discarded dry-weight analysis before volumetric analysis		
Taxon	Number	Taxon	Number
Chironomidae Simuliidae	12 5	Baetidae Leptoceridae	2 3

Code	RMWAG2	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	6.4	Weight (g)	4.5
Food Type	% Volume	Food Type	% Volume
Invertebrates Algae	100	Detritis Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae Baetidae	1 9	Chironomidae	30

Code	RMWAG3	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	9.3	Weight (g)	8
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Chironomidae Baetidae Simuliidae Leptoceridae	35 20 2 1		

Code	RMWAG4	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	10.6	Weight (g)	15.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	59	Plant material	41
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Simuliidae Chironomidae Leptoceridae	2 15 3		

Code	RMWAG5	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	9.9	Weight (g)	10
Food Type	% Volume	Food Type	% Volume
Invertebrates	83	Detritis	17
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Elmidae Leptoceridae Chironomidae Terr. Invert	2 3 5 3		

Code	RMWAG6	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	11.8	Weight (g)	19.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Simuliidae Chironomidae Trichoptera	1 4 1		

Code	RMWAG8	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	10	Weight (g)	10
Food Type	% Volume	Food Type	% Volume
Invertebrates	56	Crab	44
Notes			
Taxon	Number	Taxon	Number
Simuliidae Elmidae Baetidae Chironomidae Leptoceridae	4 1 1 2 2		

Code	RMWAG9	Date	20-10-03
Species	*A. gilli*	Site	3
Length (cm TL)	12.5	Weight (g)	21.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Simuliidae Baetidae Chironomidae Leptoceridae Trichoptera	17 15 10 2 1		

Code	RMWAG10	Date	24-04-04
Species	*A. gilli*	Site	3
Length (cm TL)	14.8	Weight (g)	31.3
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	13		
Gomphidae	1		
Leptophlebiidae	1		
Chironomidae	3		
Simuliidae	1		

Code	RMWAG11	Date	24-04-04
Species	*A. gilli*	Site	3
Length (cm TL)	12.5	Weight (g)	20.9
Food Type	% Volume	Food Type	% Volume
Invertebrates	77	Crab	22
Algae	1	Plant material	
Notes			
Taxon	Number	Taxon	Number
Potamonautidae	1		
Baetidae	23		
Simuliidae	4		
Chironomidae	1		

Code	RMWAG12	Date	24-04-04
Species	*A. gilli*	Site	3
Length (cm TL)	11	Weight (g)	18.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	5	Chironomidae	13
Baetidae	16	Simuliidae	1

Code	RMWAG13	Date	24-04-04
Species	*A. gilli*	Site	3
Length (cm TL)	10.8	Weight (g)	13.6
Food Type	% Volume	Food Type	% Volume
Invertebrates	50	Algae	50
Notes			
Taxon	Number	Taxon	Number
Trichoptera	10	Chiron. Pupa	1
Baetidae	12	Chironomidae	27

Code	RMWAG14	Date	24-04-04
Species	*A. gilli*	Site	3
Length (cm TL)	8.5	Weight (g)	6.8
Food Type	% Volume	Food Type	% Volume
Invertebrates	82	Algae	18
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	4	Baetidae	1
Elmidae	1	Simuliidae	2
Chironomidae	25		

Code	RCTAG1	Date	21-11-03
Species	*A. gilli*	Site	4
Length (cm TL)	13	Weight (g)	26
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Simuliidae	5	Chironomidae	1
Baetidae	15	Coleoptera	1
Leptoceridae	1		

Code	ACTAG4	Date	28-11-03
Species	*A. gilli*	Site	4
Length (cm TL)	10.9	Weight (g)	12.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	71	Crab	22
Algae		Plant material	6
Notes	Nematode parasites present Sand present		
Taxon	Number	Taxon	Number
Baetidae	1	Simuliidae	1
Potamonautidae	1		

Code	DCTAG1	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	12	Weight (g)	16
Food Type	% Volume	Food Type	% Volume
Invertebrates	95	Plant material	5
Notes			
Taxon	Number	Taxon	Number
Chironomidae	5	Baetidae	1
Coleoptera	1		

Code	DCTAG2	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	10.2	Weight (g)	10.5
Food Type	% Volume	Food Type	% Volume
Algae		Plant material	100
Notes			
Taxon	Number	Taxon	Number

Code	DCTAG3	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	7	Weight (g)	
Food Type	% Volume	Food Type	% Volume
Invertebrates	67	Plant material	33
Notes			
Taxon	Number	Taxon	Number
Chironomidae	5		

Code	DCTAG4	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	7.4	Weight (g)	4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		

Code	DCTAG5	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	8.1	Weight (g)	5.5
Food Type	% Volume	Food Type	% Volume
Invertebrates		Detritis	100
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number

Code	DCTAG6	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	8.2	Weight (g)	5
Food Type	% Volume	Food Type	% Volume
Invertebrates	48	Algae	52
		Plant material	
Notes			
Taxon	Number	Taxon	Number
Chironomidae	1		

Code	DCTAG7	Date	2/4/2004
Species	*A. gilli*	Site	4
Length (cm TL)	8.1	Weight (g)	5.6
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	1		
Chironomidae	5		

Code	RABAG10	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	11.8	Weight (g)	14
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes	Nematode parasites present		
Taxon	Number	Taxon	Number
Ecnomidae	3	Chironomidae	20
Heptageniidae	1	Elmidae	1
Teloganodidae	4	Ephemeroptera	1
Terr. Coleoptera	1	Chiron. Pupa	1
Simuliidae	17	Hydroptilidae	1

Code	RABAG14	Date	17-10-03
Species	*A. gilli*	Site	1
Length (cm TL)	11.3	Weight (g)	17
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Elmidae	2	Baetidae	25
Simuliidae	16	Ephemeroptera	1
Teloganodidae	11	Chironomidae	30
Trichoptera	1	Leptoceridae	1

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e) *Micropterus dolomieu*

Code	RRDMD1	Date	22-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	12.5	Weight (g)	23.5
Food Type	% Volume	Food Type	% Volume
Crab	100	Fish	
Notes			
Taxon	Number	Taxon	Number
Potamonautidae	1		

Code	RRDMD2	Date	22-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	9.2	Weight (g)	14
Food Type	% Volume	Food Type	% Volume
Crab		Fish	
Notes	1 insect-like root fragment		
Taxon	Number	Taxon	Number

Code	RRDMD3	Date	22-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	12.8	Weight (g)	27.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	3	Chironomidae	1
Terr. Diptera	1	Baetidae	1

Code	RRDMD4	Date	29-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	15	Weight (g)	36.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Terr. Coleoptera	1		
Terr. Inv	1		

Code	RRDMD5	Date	29-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	13.8	Weight (g)	29.1
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Ephemeroptera	1		

Code	RRDMD6	Date	29-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	10.5	Weight (g)	13.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	11	Corixidae	1
Baetidae	18	Simuliidae	1
Terr. Inv	1		

Code	RRDMD7	Date	29-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	9	Weight (g)	11.8
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	1		
Baetidae	6		

Code	RKUMD1	Date	22-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	10.9	Weight (g)	13.4
Food Type	% Volume	Food Type	% Volume
Invertebrates	99	Plant material	1
Notes			
Taxon	Number	Taxon	Number
Gomphidae	1	Coleoptera	1
Coenogrionidae	7	Dytiscidae	1
Culicidae	1	Baetidae	1

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Code	RKLMD1	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	13.4	Weight (g)	22
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Aeshnidae	1		

Code	RKLMDL1	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	3.4	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Baetidae	18		

Code	RKLMDL2	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	2.9	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	12		
Chironomidae	1		

Code	RKLMDL3	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	3	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	12		

Code	RKLMDL4	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	3	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	25		

Code	RKLMDL5	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)		Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	4		

Code	RKLMDL6	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	3.2	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	25		

Code	RKLMDL7	Date	22-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	4.7	Weight (g)	Too light
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Baetidae	12		

Code	ARDMD1	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	9.5	Weight (g)	13
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	3		
Leptophlebiidae	1		
Baetidae	20		

Code	ARDMD2	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	10.7	Weight (g)	16
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Lepidoptera	1	Baetidae	15
Potamonautidae	1	Chironomidae	1
Heptageniidae	5	Ephemeroptera	1

Code	ARDMD3	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	11.2	Weight (g)	18
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	1		
Baetidae	10		

Code	ARDMD4	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	12	Weight (g)	21
Food Type	% Volume	Food Type	% Volume
Invertebrates	91	Detritis	
Algae		Plant material	9
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	1	Libellulidae	1
Coenogrionidae	2	Baetidae	10

Code	ARDMD5	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	No record	Weight (g)	38
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Ecnomidae	3	Heptageniidae	1
Baetidae	5	Gerridae	1
Terr. Invert.	1	Terr. Coleoptera	1
Gyrinidae	1	Terr. Hemiptera	1

Code	ARDMD6	Date	23-10-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	13.2	Weight (g)	27
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Baetidae	29		
Diptera Larva	4		
Corydalidae	2		
Ecnomidae	3		

Code	ARDMD7	Date	27-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	26.8	Weight (g)	237
Food Type	% Volume	Food Type	% Volume
Invertebrates	30	Detritis	
Crab	70	Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Potamonautidae	1		
Terr. Invert.	3		
Notonemouridae	1		

Code	ARDMD8	Date	27-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	13	Weight (g)	23.2
Food Type	% Volume	Food Type	% Volume
Invertebrates	97	Detritis	
Crab		Fish	
Algae		Plant material	3
Notes			
Taxon	Number	Taxon	Number
Heptageniidae	2		
Baetidae	2		
Terr. Diptera	12		

Code	ARDMD9	Date	27-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	10.8	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Potamonautidae	1		
Ecnomidae	1		
Terr. Invert.	1		
Lepidoptera	1		

Code	ARDMD10	Date	27-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	12.6	Weight (g)	22.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	92	Detritis	8
Notes			
Taxon	Number	Taxon	Number
Ecnomidae	6	Potamonautidae	1
Gerridae	2	Baetidae	1
Terr. Hemiptera	1	Heptageniidae	2
Terr. Diptera	1	Ephemeroptera	1
Corixidae	2		

Code	ARDMD11	Date	27-11-03
Species	*M. dolomieu*	Site	5
Length (cm TL)	13	Weight (g)	24.7
Food Type	% Volume	Food Type	% Volume
Invertebrates	45	Detritis	17
Crab		Fish	37
Notes			
Taxon	Number	Taxon	Number
Fish	1	Ephemeroptera	7
Dytiscidae	2	Terr. Hemiptera	1
Heptageniidae	1		

Code	ARDMD12	Date	29-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	15.5	Weight (g)	41.9
Food Type	% Volume	Food Type	% Volume
Crab	98	Plant material	2
Algae		Fish	
Notes			
Taxon	Number	Taxon	Number
Potamonautidae	1		

Code	ARDMD13	Date	30-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	14.5	Weight (g)	35.5
Food Type	% Volume	Food Type	% Volume
Invertebrates	97	Detritis	3
Crab		Fish	
Algae		Plant material	
Notes			
Taxon	Number	Taxon	Number
Libellulidae	1	Trichoptera	4
Corixidae	10	Baetidae	4
Terr. Invert.	1		

Code	AKLMD1	Date	27-11-03
Species	*M. dolomieu*	Site	7
Length (cm TL)	32.5	Weight (g)	393
Food Type	% Volume	Food Type	% Volume
Invertebrates	0.5169	Detritis	0.5169
Crab	0.7906	Fish	
Algae		Plant material	
Notes	stomach contents used in for discarded dry-weight analysis		
Taxon	Number	Taxon	Number
Potamonautidae	2		
Libellulidae	3		
Terr. Invert	2		

Code	ARDMD14	Date	30-04-04
Species	*M. dolomieu*	Site	5
Length (cm TL)	No record	Weight (g)	No record
Food Type	% Volume	Food Type	% Volume
Invertebrates	100	Detritis	
Notes			
Taxon	Number	Taxon	Number
Trichoptera	2		
Baetidae	1		

Appendix 3: Photographs

a) Study Sites

Plate 1: Site 1 "Algeria"

Plate 2: Site 2 "Rockface"

Plate 3: Site 3 "Meadow"

Plate 4: Site 4 "Cottage"


Plate 5: Site 5 "Rooidraai"


Plate 6: Site 6 "Upper Keurbos"

Plate 7: Site 7 "Lower Keurbos"

Plate 8: Site 8 "Canyon"


Plate 9: Extra Site A "Jamaka"


Plate 10: Extra Site B "Keurbos extra riffle"

b) Barriers to fish movement

Plate 11: The Algeria causeway

Plate 12: The waterfall barrier upstream of Site 5

Plate 13: The abstraction weir upstream of Clanwilliam dam