RIVER REHALIBITATION: LITERATURE REVIEW, CASE STUDIES AND EMERGING PRINCIPLES

by

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EXECUTIVE SUMMARY

E1. Introduction

Land and water developments worldwide have brought many benefits to humans, but also have led to a decline in the ecological form and functioning of rivers. There is growing realization of the national costs of poorly functioning rivers through, for instance, the decline of fisheries, the increasing need for expensive water-quality treatment, sedimentation of reservoirs, increasing severity of floods and flood damage, the need for expensive flood-control measures, the loss of recreational and biodiversity values, and much more. Because of this, rehabilitation of rivers is moving from a background of 'grey' science to a phase of increasing structure with a rapid build up of data and knowledge.

This three-year project was planned to contribute toward the development of a South African pool of expertise on river rehabilitation.

E2. Background to the study

The Water Research Commission awarded separate contracts to the Freshwater Research Unit, University of Cape Town (UCT) and the Department of Earth Sciences, University of the Western Cape (UWC) to jointly run the project. Presented here is the joint final report.

The overall approach was designed to:

- create formal research links between fluvial geomorphologists at UWC and freshwater ecologists at UCT;
- begin development of expertise in the field of river rehabilitation.

Through a process of discussion with the steering committee, the project focused on the three following objectives:

- review of the world literature on river rehabilitation;
- completion of river-response studies on three Western Cape rivers where active river management was underway;
- derivation of a first draft of geomorphological and ecological principles for river rehabilitation.

This report is divided into four parts.

1. Introduction
2. The literature review
3. The river studies
4. Conclusion, including derivation of emerging principles

The project focused on physical aspects of river degradation and rehabilitation, whilst acknowledging that many other aspects, such as water chemistry and the legislative framework, would have to be addressed as part of a comprehensive strategy of river rehabilitation.

E3. The literature review

Introduction

There is no consensus internationally about what constitutes *river rehabilitation* or *river restoration*. The two terms are used loosely to describe a variety of projects with different goals. The ambiguity is largely a result of the value that different people place on the natural environment and what constitutes a natural environment. Proponents of the Balance of Nature paradigm believe restored river systems should be protected from outside (non-natural) human forces that change ecosystems, whilst those of the new Flux of Nature paradigm view ecosystems as continually disturbed, both by natural and anthropogenic forces, and responding to differing levels and different kinds of disturbance. In their view the biological constituents present in an ecosystem will not necessarily be the same as those in the future. Whichever view is held, there is a common goal of desiring to return a degraded river ecosystem to a more natural state.

The underlying assumption behind almost all river rehabilitation projects has been that the re-introduction of a variety of physical habitats will improve the ecological functioning of a degraded ecosystem. Degradation of the river, however, could take many forms, and physical rehabilitation may not always be necessary or appropriate.

Degradation of aquatic ecosystems

Physical disturbance

Physical disturbance of the river can result from human interventions either within the catchment or the channel. Two common channel disturbances are channelisation and canalisation. Channelisation involves straightening, deepening, widening or narrowing the channel, maintaining the banks in some desired (unnatural) form, or erecting embankments. Canalisation is an extreme form of channelisation, whereby the channel bed and banks are lined with concrete. The ecological repercussions of channelisation are many and complex, with a loss of heterogeneity that inevitably translates into a loss of ecological biodiversity.

Hydrological manipulations

Humans have intervened in all land-based parts of the hydrological cycle. Whilst dams represent the greatest intervention to rivers, inter-basin transfers of water (IBTs), direct abstraction, agricultural activities and urbanisation each have their own unique impacts. In general, these interventions have changed the pattern of high and low flows and sediment regimes of rivers, which have in turn led to changes in channel morphology, water quality and the biota. Abstraction rates from groundwater are also rapidly increasing worldwide. Generally, withdrawals from groundwater aquifers result in a lowering of the water table, a consequent reduction in stream flow, possibly with an intrusion of saline water, and so a decrease in water quality. The ecological effects of groundwater abstraction on riverine ecosystems are not well documented.

Chemical disturbance

Water quality naturally varies between rivers due to climate and the underlying geological formations, and through the year due to seasonal changes in flow and temperature and responding biological activity. Humans impact on water quality and quantity through changing drainage patterns to rivers and the chemical constituents and sediments the water may be carrying. They also use rivers as wastewater conduits and feed urban storm-water drains into them. Pollutants impact aquatic ecosystems by reducing abundances and biodiversity, and their general ability to function efficiently. Rehabilitation of degraded systems requires that attention be paid to the quality of the water, as well as the quantity and the physical attributes of the channel. Unless all three are suitable, planned rehabilitation may be unsuccessful.

Planning rehabilitation

Planning rehabilitation

A more holistic approach to river rehabilitation is evolving. This includes specialist inputs from relevant groups including government agencies, local municipalities, river rehabilitation agencies, financial institutions, civil and hydraulic engineers, freshwater ecologists, palaeo-ecologists, historical geographers, fluvial geomorphologists, water and navigational agencies, operation companies, the general public, riparian landowners, and educational and research institutes. Adopting a holistic approach to rehabilitation adheres to the concept of Integrated Catchment Management, whereby the whole catchment is considered and not just the reach to be rehabilitated. Successful rehabilitation projects follow nine main steps. These are: developing goals and measurable objectives; river condition analysis; prioritization and project constraints; strategies; study design, techniques and rehabilitation alternatives; feasibility studies; implementation; monitoring and maintenance; and post-project evaluation.

Assessing river condition

Assessing river condition is an essential step in rehabilitation projects. The most useful assessments are those that compare the existing measured condition of the river to its natural condition, or to its prior state if that is less than natural, in order to measure how degraded or close to natural the system is. This prior, or reference, condition can be used to inform debate about the desired condition of the river and eventually to set the rehabilitation goals.

Rehabilitation

Implementation – methods and materials

Successful rehabilitation efforts are based on three main activities or measures. These are environmentally sensitive river maintenance activities, soil bioengineering or soft biotechnical engineering practices and measures to mitigate the effects of hard engineering practices. Natural recovery of the river channel, without intervention, is an option where time and space allow and where other limitations, such as excessive water abstraction, are not present.

Rehabilitation of banks, riparian zones and floodplains

A range of off-channel rehabilitation activities can be part of rehabilitation, such as reconnection of floodplains to rivers, re-instatement of meanders, stabilisation of stream banks, and rehabilitation of riparian zones. Riparian zones can be introduced as buffer strips offering protection from catchment activities, and artificial wetlands can aid management of water quality. Five methods of stabilising stream banks are: 1) vegetation alone; 2) vegetation with structural control; 3) vegetation and structural control with bank shaping; 4) structural control alone and 5) bioengineering methods.

Interventions in the channel

The goal of intervening within the channel is to re-create habitat diversity and velocity variation. Rehabilitation of instream habitat will only work if the limiting factor affecting recovery at the site is habitat simplification. Common factors affecting the success of introduced structures are the movement of sediments, the scouring potential of the water, and the flow regime. Three categories of instream structures used to manipulate and facilitate changes to aquatic habitats are cross-channel structures, partially intruding structures and fishways.

Monitoring and post-project appraisal

Monitoring and post-project evaluation differ. Monitoring involves regular or continuous measurement of selected ecosystem variables. Evaluation is a sub-activity of a larger monitoring programme, whereby collected data on various parameters are compared to pre-determined goals for those parameters to assess efficacy of the rehabilitation effort. There have been few post-project evaluations of aquatic rehabilitation projects. Those documented reveal a high percentage of failures. Lack of post-project evaluation is detrimental to the development of ‘restoration ecology’ as a science.

The river-response studies

The Lourens River

The river rises on the Hottentot-Hollands mountains, and flows into False Bay through Somerset West. There are pine plantations, orchards, prestigious wineries and a piggery along its middle reaches, and an urban area on its lower reaches. The river has been the subject of many local rehabilitation efforts, and is a proclaimed Protected Natural Environment. The original plans to document the river’s response to clearance of alien vegetation had to be cancelled, as did the next plan to similarly document river response to flood control measures in the urban area, both because of changes of plans by other organisations. Finally, in the second year of the project, this study centred on a geomorphological investigation of two similar reaches of the river, their hydraulic characteristics and their responses to floods.

The following conclusions were drawn.

- Channel morphology was largely determined by high flows.
- Hydraulic biotopes changed as discharge changed.
- Aquatic invertebrate assemblages indicated the river ecosystem was in a fair to good condition.
- Floods enhanced the diversity of riparian and aquatic plants.
- Channel processes such as degradation and aggradation, and the occurrence of aquatic organisms, could be correlated with the distribution of different sized sediments.

The Kuils River

The river rises on the Cape Flats, and once probably flowed only during the rainy season. Now, water-treatment plants discharge treated sewage into it, and it is sluggishly perennial. It is highly impacted, as it flows through both formal and informal settlements, industrial areas and semi-rural areas. Alien trees are being cleared from its lower reaches. The study focused on engineering works on the middle reaches of the river within the urban environment of Kuils River. It was concluded that the Kuils River provided a classic example of the impacts of channelisation on the geomorphological and ecological nature of a river, as follows.

Geomorphological impacts

- Channel widening and straightening resulted in a uniform channel shape.
- The diversities of substrata and flow types were reduced.
- Channelisation was followed by an upstream migration of bank erosion, which in turn, was followed by massive bank collapse during the wet season.

Ecological impacts

- The geomorphological changes resulted in reduced available aquatic habitats.
- This in turn resulted in low abundances and diversities of aquatic invertebrates.
- The engineering construction phase caused soil disturbances and a resulting increased diversity in plant species, most of which were alien annual weeds.

Impacts on infrastructure

- Upstream migration of bank erosion threatened the support structures of the Old Bottelary Road.

The Silvermine River

The river is about 12 km long, rises on Table Mountain and drains into False Bay. All of its catchment has recently been taken over by National Parks except for the last 1-2 km that runs through municipal land, a golf course, and then the estuary. Much of the upper catchment burnt in the January 2000 fires, and the remaining alien vegetation is being hand-cleared. The study focused on responses of the river to the 2000 fires and to clearance activities at different points along its length. National Parks policy is to not intervene in the natural recovery of the river other than through clearance of the alien plants.

It was concluded that different sections of the catchment were responding in different ways to the fires and clearing methods. Recovery of indigenous vegetation was more successful in old pine areas than in those that supported *Acacia*. The riparian and in-stream plant communities were responding more rapidly than that of the hillslopes. It seemed likely that recovery of riverine habitats would occur successfully unaided, if alien clearance continued and as long as the river corridor was guarded from unnecessary disturbance. This would require maintaining a clear boundary zone alongside the river with no stacking and burning of debris. One exception to this general policy of minimal intervention might be the middle reaches of the river, where the river was eroding through a large ancient deposit of sand. Near vertical sandy banks collapsed into the river with every flood, affecting all downstream reaches. More investigation of this deposit, including consideration of possible active removal, is recommended.

Conclusion

Emerging geomorphological and ecological principles

The science of river rehabilitation is widely seen as still in a pre-paradigmatic state with a weak conceptual basis to guide the plethora of procedures being undertaken. Where principles are given, the focus is mainly on a wider set of management principles, such as ensuring that objectives are realistic in terms of budget, or that stakeholders are involved. Such principles are vital for a successful rehabilitation effort, but nested within them should be a sub-set of ecological and geomorphological principles that will guide the scientific aspects of the work. Without these, rehabilitation efforts stand the risk of being random experiments that make little or no contribution to scientific advancement or technical expertise.

What appear to be some emerging scientific principles are given below. It is recognised that many more may appear with time, and several could eventually be refined or become part of more encompassing principles. The wider management principles are not addressed here, nor are the ways principles may have to modified due to the constraints of human activities within the catchment.

General scientific principles of river rehabilitation

- Rehabilitate back toward natural
- Treat causes rather than symptoms
- Rehabilitation is an interdisciplinary activity
- All major abiotic drivers of ecosystem processes should be in harmony with the rehabilitation objectives
- Natural recovery may not achieve the set objectives
- It is easier to cross a degradation threshold than to return over it
- Ecosystems are dynamic and can naturally exist in alternative metastable states
- Monitoring is an essential component of rehabilitation
- Rehabilitation operates over a range of spatial and temporal scales

Principles at the landscape level

- Catchment-level processes affect local form and function
- Assessment and planning should start at the catchment scale
- Rehabilitation should progress from source to mouth, and include tributaries

Geomorphological principles

- Channel features are a product of their catchments – work with the river not against it
- Different river zones have different abilities to mobilize sediments – let the river do the work
- Dynamic channels are more natural than restricted ones
- Straightening and smoothing channels increases erosive power
- A range of different magnitude flows is required to maintain a channel
- Floodplains, wetlands and groundwater are part of the river ecosystem and should be included in rehabilitation projects
- Sites with all their potential geomorphic units offer the optimal diversity of aquatic habitat

Ecological principles

- Biotic as well as abiotic factors determine the communities that a river can support
- Maximizing habitat for one species is not the same as re-creating the biotic structure and functioning of a stream ecosystem
- Re-colonisation of damaged areas will occur at levels dependent on the habitats available, stocking sources and remaining stressors
- The rate of recovery is dependent on the ecosystem and the intensity of the disturbance

Conclusions and recommendations

River rehabilitation is characterised by a dearth of testable hypotheses, and consequently an absence of rehabilitation principles. As a new science, its development depends on accumulating empirical observations, using these to develop and test hypotheses, and from that tentatively beginning the development of principles. In this WRC project, it was possible to follow part of that sequence, by empirical observation of how three rivers responded to management interventions. A review of the international literature allowed further understanding, by linking some of these observations to wider rehabilitation knowledge. The principles presented in Chapter 15 could not emerge from this via hypothesis testing because it was not possible to set up such a research programme within the financial, monetary and management constraints faced. Thus, the hypothesis-testing phase was missed, and the principles were instead drawn from an understanding of river ecosystems gained before and during the project and from scattered information in the international literature. This is a first attempt to present a set of principles as some relevant over-arching wisdoms, and it became clear that the list could be almost any length. Time and testing will prove in an iterative manner which of the principles are valid, which are not particularly relevant or true, and what is missing.

There are a multitude of rehabilitation projects being undertaken worldwide, each of which could be viewed as an experiment from which considerable knowledge could be obtained. Each such experiment could be: designed to test one or more hypotheses; objectively documented; evaluated; and the results of the evaluation widely disseminated. This would greatly aid the structured development of a set of rehabilitation principles.

To support this process, a co-ordinated programme of gathering ecological and geomorphological data on rehabilitation is needed that will provide the theoretical basis for formulation and testing of hypotheses. This is particularly important because river response to intervention is slow and so understanding will grow slowly anyway. Such a programme would be most effective if run over a number of countries with a central co-ordination centre, and in its simplest form could perhaps merely provide a means of guiding rehabilitation exercises in a way that allows hypothesis testing to take place. Monitoring could then be used to assess the validity of the hypotheses, with the results fed through to the central point responsible for principles development. Managers would benefit though a structured growth of understanding and guidelines for rehabilitation.

The following recommendations are made.

- Create a centre of expertise on river rehabilitation within South Africa.
- Link up with similar groups globally to develop hypothesis-testing rehabilitation projects.
- Provide structured funding for rehabilitating rivers and evaluating outcomes.
- Develop a scientific research programme for river rehabilitation.
- Synthesise findings of the research and rehabilitatitive activities and further develop a set of rehabilitation principles.
- Develop practical guidelines from the principles.
- Develop close liaison between scientists and managers.
- Develop appropriate policy and regulatory frameworks.

Capacity building

Individuals

- Lindie Smith (Chapter 12) and Ruth-Mary Fisher (Chapter 13) at UWC are each registered for an MSc and will submit theses during 2003.
- Karl Reinecke (Chapter 14) at UCT will register for an MSc in 2003.
- Alice Williams (UWC) and Dave Smith, Barry O'Donoghue (UCT) learnt skills in field data collection and data entry and analysis.
- Edward Akunji (UCT) completed a B.Sc. (Hons.) thesis on a topic guided by the Silvermine River study.
- Ian Bowker, Colin Logan and Jateen Bhana (UCT) completed fourth-year undergraduate engineering projects on the three study rivers.
- Anna Cubison (University of Newcastle, England) completed a B.Sc (Hons.) thesis on the Silvermine River: *Comparison of the social and economic impacts of river rehabilitation in Britain (using the example of the River Dearne) with a South African river rehabilitation project.*

Institutional

Neither the fluvial geomorphologists at UWC or the freshwater ecologists at UCT had experience in river rehabilitation before this project. Both groups have some competence in the field now and intend to continue research into river rehabilitation. Science-management links strengthened during the project through collaboration with National Parks, the City of Cape Town and DWAF Working for Water.

Community

At the request of the South Peninsula Administration of the City of Cape Town (CCT), Karl Reinecke made a presentation on the Silvermine River to the local community, and completed a brief study of its surface water hydrological network for CCT.

ACKNOWLEDGEMENTS

Two senior members of the project left the University of the Western Cape during its lifetime. Professor Jan Boelhouwers and Dr Steve Holness are thanked for their contributions.

Three national advisors and one international advisor guided aspects of the project and are thanked for their valued advice and guidance. The national advisors were Dr Neil Armitage, Department of Civil Engineering, University of Cape Town; Dr Charlie Boucher, Botany Department, University of Stellenbosch; and Prof. Kate Rowntree, Geography Department, Rhodes University. All visited all three study rivers on at least two occasions and the team benefited greatly from their inputs. The international advisor was Prof Malcolm Newson of the University of Newcastle-upon-Tyne, U.K. with whom we shared a most informative visit to the Silvermine sites and who provided input on the nature of the channel at Site 4. Other specialists who visited the Silvermine site and discussed aspects of project design and interpretation of results with us were Dr Mike Acreman and Dr Douglas Booker, Centre for Ecology and Hydrology, Wallingford, U.K.; and Dr Tony Ladson, Dept of Environmental Engineering, University of Melbourne, Australia.

M.C. Briers and Pieter Bornman, the Department of Geomatics, UCT, did all the surveys of cross-sections and produced the channel maps.

For the Kuils and Lourens studies, the two researchers - Ms Smith and Ms Fisher – worked as field assistants for each other. Alice Williams, field assistant, Department of Earth Science, UWC, who helped with fieldwork on the Lourens, entered the vegetation data into Turbo Veg and entered the references for the literature review into Reference Manager. Mr. Frans Weitz and Dr. Lincoln Raitt, Department of Biodiversity and Conservation Biology, UWC, identified the plants for the Kuils and Lourens studies and helped with the interpretation of the vegetation results. Mike Luger, Senior Environmentalist and Rudolf de Haan, Senior Engineer, at Ninham Shand Consulting, provided information on channelisation of the Kuils and the situation prior to channelisation.

Plant identifications were done for the Silvermine sites by Teri Trinder-Smith, Bolus Herbarium, UCT. Also at UCT, Dave Smith and Barry O'Donoghue worked hard and cheerfully as field and laboratory assistants for the Silvermine study, processing and collating data. Ian Bowker (Department of Civil Engineering, UCT) completed a fourth-year project on the hydraulics of the Silvermine system, to inform the study of that river, and Edward Akunji (Environmental and Geographical Sciences, UCT) completed an Honours project on the sand plug at Site 4 on the Silvermine River, providing new insights into its origin.

Field assistants working on the Silvermine study, other than project staff and those mentioned above, are thanked for their invaluable hours of hard work in the sun: Sean Tindale, Robbie Blain, Gareth Sleigh, Rachel Wilson, Katharina von Gerhardt, Dave Miller, Troy Girdler-Brown, Carlos Sebastian and Christine Jarvis.

The city of Cape Town funded the collection of hydrological data for the Silvermine catchment. A special thanks to Julia Wood and Bill Harding for key roles in this collaboration.

Finally, the project team thanks most sincerely for their wise guidance Drs Sizwe Mkhize and Steve Mitchell of the Water Research Commission, and our Steering Committee: Dr Neil Armitage, Dr Charlie Boucher, Mr John Dini, Mr Dana Grobler, Ms Angelina Jordanova, Dr Heather Malan, Mr Mark Obree, Prof. Kate Rowntree, Dr Cornelius Ruiters, Ms Christa Thirion, Dr Mandy Uys and Ms Julia Wood.

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1. INTRODUCTION

1.1 Rationale for the project

Land and water developments worldwide have brought many benefits to humans, but have also led to a decline in the ecological form and functioning of rivers. There is growing realization of the national costs of poorly functioning rivers through, for instance, the decline of fisheries, the increasing need for expensive water-quality treatment, sedimentation of reservoirs, increasing severity of floods and flood damage, the need for expensive flood-control measures, the loss of recreational and biodiversity values, and much more. Because of this, rehabilitation of rivers is increasingly being attempted, moving from a background of ‘grey’ science to a phase of increasing structure with a rapid build up of data and knowledge.

River rehabilitation as a science is in its infancy in South Africa. Many engineering groups are involved in ‘improvement’ of degraded rivers and, in the last one to two decades, river scientists have often also been involved. Probably their earliest structured involvement in river rehabilitation in South Africa was in the field of environmental flow assessments, where they developed methods to advise on the ecological and subsistence consequences of manipulating flow regimes (King & Louw 1998; King *et al.* 2000). Although this advice mostly related to the impacts that could be expected with new water-resource projects, in some cases, flows with some semblance to natural states were designed for rehabilitation of rivers degraded by earlier flow losses. Rehabilitation of water chemistry is in its early stages, with documents such as Dallas & Day (1993), Pegram & Görgens (2001) and Malan & Day (2002) providing information on the links between chemistry and ecosystem functioning upon which chemical rehabilitation could be based. Similarly, knowledge is accumulating on the physical nature of channels and hydraulic habitats within them, from such work as Rowntree & Wadeson (1999), Jonker *et al.* (2001) and King & Schael (2001).

This three-year project was intended as an early initiative, building on these past projects, to begin development of understanding and expertise in the science of river rehabilitation.

1.2 Background to the project

The original proposal emanated from an approach by Prof Jan Boelhouwers of the University of the Western Cape (UWC) to Dr Jackie King of the University of Cape Town (UCT) regarding a possible collaborative research project. It was suggested that fluvial geomorphologists from UWC could work with freshwater ecologists from UCT to develop experience in the field of river rehabilitation.

The proposal submitted to and approved by the Water Research Commission was thus a joint one from the two universities, and centred on a project with two overall aims: to create formal research links between geomorphologists at the University of the Western Cape and freshwater ecologists at the University of Cape Town, and to jointly begin development of expertise in the field of river rehabilitation. In this initial exploratory project, most activities would focus on becoming familiar with what has been done elsewhere in this field, and on developing a “rehabilitation mindset” and a common language and understanding within the team.

The proposal was that the project team would become involved in actual rehabilitation projects already underway in one rural river and one urban river. Other activities in the proposal included a literature review, creation of a reference collection of international literature, and the derivation of geomorphological and ecological principles for river rehabilitation. The Lourens River was chosen as the rural river, as Working for Water (WfW) teams of the Department of Water Affairs and Forestry were clearing alien trees from the banks of the middle and upper reaches. The proposal for the Lourens thus was to record their activities and clearance methods, and track their success and the river's response. No urban river was chosen at that stage, mainly because alien clearance of the Lourens halted shortly after the start of the project and was not planned to begin again for at least a year, and so the whole project had to develop a new focus.

In a change of plans approved by the Steering Committee on 20 November 2000, the focus changed to sections of three rivers where management activities were underway. Two researchers at UWC and one at UCT were each allocated one of the rivers, where they would record the management interventions and the nature and response of the river, as follows.

- <u>UWC Project: the Lourens River (Chapter 12).</u> The river rises on the Hottentot-Hollands mountains, and flows into False Bay through Somerset West. There are pine plantations, orchards, prestigious wineries and a piggery along its middle reaches, and an urban area on its lower reaches. The river has been the subject of many local rehabilitation efforts, and is a proclaimed Protected Natural Environment. As well as the alien clearance by WfW, flood-control measures were underway in the urban area of Somerset West and these would be one focus of the project.
- <u>UWC Project: the Kuils River (Chapter 13).</u> The river rises on the Cape Flats, and once probably flowed only during the rainy season. Now, water-treatment plants discharge treated sewage into it, and it is sluggishly perennial. It is highly impacted, as it flows through both formal and informal settlements, industrial areas and semi-rural areas. Aliens are being cleared from its lower reaches. The study would focus on flood control works on the middle reaches of the river within the urban environment of Kuils River.
- <u>UCT Project: the Silvermine River (Chapter 14).</u> The river is about 12 km long, rises on Table Mountain and drains into False Bay. Almost all of its catchment has recently been taken over by National Parks, with only the last 1-2 km running through municipal land, a golf course, and then the estuary. Much of the upper catchment burnt in the January 2000 fires, and remaining alien vegetation is being hand-cleared. The study would focus on responses of the river to clearance activities at different points along its length.

For each river, the main data-collection activities would extend over the summers of 2000/2001 and 2001/2002, with additional investigations to assess responses of the rivers to the winter 2001 floods. As the first summer of fieldwork began, the flood-control measures planned for the lower Lourens were halted, with no clarity on when and if they would re-commence. Thus, for the second time, the research plans for the Lourens had to be changed. Because the research project was at that stage already in its second year and a summer of baseline data had been collected, the new research plan had to become independent of interventions planned by local authorities. Instead, the site where data had been collected (some of which were now irrelevant) was paired with an upstream site in the same zone but with different reach and site characteristics, for a comparative study of channel morphology and local hydraulics.

Difficulties were also experienced in the two other studies, which probably reflect typical problems encountered in research projects of limited duration in areas subject to a range of management activities. In the Kuils, for instance, uncertainty over the extent of planned engineering works and their onset led to some data collection having to be repeated, and study sites being established with no certainty of whether or not they would prove relevant or even exist a year later. In the Silvermine, the clearance of alien trees in an *ad hoc* programme as funds and contractors became available, led to different parts of the riverbanks and catchment being cleared at different times and the river being affected in different ways, all of which were superimposed on the scientific study within this project. In all three programmes, considerable time and effort was invested in simply trying to keep up to date with the plans of local authorities, and ensure that planned research activities were still relevant.

In parallel with the field studies, a review of the international literature was completed and a compatible reference database set up at each university. The UCT database is stored in ENDNOTE and the UWC database in REFERENCE MANAGER. At project end, a copy of each database will be held at each university, and a reference may be located at one of the universities by searching both databases with either of the software packages.

1.3 Project objectives

In summary, the project objectives, as redefined through the steering committee meetings were:

- write a review of the world literature on river rehabilitation, including initial problem conditions and remedial actions;
- complete pilot studies of river response on three Western Cape rivers where management activities are taking place;
- derive a first draft of emerging geomorphological and ecological principles for river rehabilitation.

The project focus was on physical aspects of rivers and rehabilitation. It is acknowledged that other aspects such as water quality, and legal and management aspects of rehabilitation are important, but these remain outside the focus of this report. Readers should also bear in mind that the literature review presented here is a first attempt to summarise the world literature on this large and diffuse subject. It is intended that the review should continue through subsequent projects.

1.4 Project report

The report consists of four parts. Following this introductory part, Part 2 consists of a ten-chapter literature review and Part 3 covers the three field studies. Part 4 represents a conclusion, with a chapter on emerging geomorphological and ecological principles for rehabilitation, and one providing conclusions and recommendations.

2. INTRODUCTION TO LITERATURE REVIEW

2.1 The river ecosystem

In this document, the term *river* or *river ecosystem* is used as an all-encompassing term to describe the living and non-living parts of the river channel, its banks, connected seeps, wetlands, lakes, floodplains, deltas and groundwater, and the estuary. The main non-living, or abiotic, components of the river ecosystem are thus (with the relevant discipline):

- the flow regime of, or pattern of flow in, the river (hydrology);
- the behaviour of the water as it flows through the system (hydraulics);
- the movement of large and small particles along the system (sedimentology);
- the interactions between these loads and flow to create different kinds of channel and bank features (fluvial geomorphology);
- the chemical, physico-chemical and thermal characteristics of the water (aquatic chemistry).

The main living, or biotic, components of the river ecosystem are:

- aquatic, marginal and riparian plants, including riverside trees;
- fish;
- aquatic invertebrates;
- aquatic mammals;
- the herpetofauna;
- the microfauna and flora;
- water birds;
- river-dependent terrestrial wildlife.

Current ecological thinking on rivers recognises the inter-connectedness of all of these parts of the river system, and thus the potential that change in one component or area can affect another. Indeed, rivers are uniquely linear ecosystems, functioning in an inter-linked way from source to estuary, and are affected by most human activities in the catchments they drain. Reflecting the landscape, climate and geological nature of their catchments, both their abiotic and biotic nature changes from source to estuary. Geomorphological zones, such as *mountain, foothill,* or *lowland* (Rowntree & Wadeson 1999) are reflected in different chemical and thermal regimes (e.g. Dallas 2002) and the presence of characteristic plant and animal communities (e.g. King & Schael 2001). Upstream reaches dictate the flow of sediments, water, and suspended and dissolved substances including nutrients, to downstream reaches, influencing the nature of the lower reaches. Disturbances in the upper reaches inevitably impact downstream reaches to a greater or lesser extent. In many ways, downstream reaches may also influence upstream reaches, for instance, through fish migrations and up-valley flight of emerging aquatic insects.

Flow and topography are primary determinants of the nature of rivers, dictating erosion and deposition patterns and sediment transport, and thus sculpting the physical environment of channel and banks. The higher reaches tend to be steeper and occur in regions of hard geological strata, and so the water flows faster in narrow confined channels with coarse substrata. Floodplains, and wetlands (other than mountain seeps), are rare and riparian belts of trees are narrow. Lower and/or flatter areas have slower flow, finer substrata, wider valleys and more potential for the development of wetlands, floodplains and deltas. Local hydraulic characteristics differ through all these changes of topography and substrata, creating mosaics of different substratum-flow conditions that offer a variety of habitats for plants and animals. Providing other conditions (such as water chemistry, water temperature, lack of alien invader species) are favourable (Section 2.3.3), the greater the diversity of habitat the greater the potential for high biodiversity.

Different parts of the flow regime are important for different components of the ecosystem. Low flows maintain a basic wetted area, providing year-round aquatic habitat without which species such as fish could not be present. Different levels of low flow in the dry and wet season provide a range of habitats along the margins of channels. This allows many vegetal species to establish in a sequence of zones up the shoreline where levels of inundation and exposure through the year are suitable. These vegetation zones in turn provide a range of habitats that support many animal species.

Small to medium floods may be responsible for much of the sorting of coarser substrata (gravels, cobble), providing physical heterogeneity that is lost in rivers where these floods are harnessed by dams (King & Schael 2001), with a consequent loss of biodiversity. All floods, from small intra-annual ones to the largest inter-annual ones, influence the distribution pattern of riparian tree species. For instance, the tree line along many southern African rivers appears to be linked to the 1:2 year flood line; the outer edge of the riparian zone to a flood with a return period of more than 50 years; and the wetbank vegetation (sedges, reeds, ferns, mosses and allied species) to the larger of the intra-annual floods (Boucher 2002; King *et al*. in press).

Floodplains, secondary channels and other kinds of wetlands are also linked to specific size ranges of floods, experiencing cycles of wetting and drying that provide rich inputs of carbon, nutrients and micro-organisms to the river. At the same time, the high levels of primary productivity and the quiet waters in these wetlands lead to a proliferation of micro-organisms, invertebrates and larger animals such as fish. Many of these animal species move from the river to complete life-cycle stages in the wetlands. The interval between wetting and drying cycles dictates whether these species can complete these life stages successfully. This interval also influences how much organic material accumulates in the wetlands; too-frequent wetting results in little accumulation of organic material and inadequate wetting in accumulated material rarely entering the river channel.

The nature of many river ecosystems appears to be dictated by, and dependent on, these regular inputs from wetlands. Together, the channel and its wetlands provide a network of areas that is inter-connected longitudinally and laterally. Both kinds of links are important in the energy and nutrient dynamics of the ecosystem, and in the reproduction and maintenance of species.

The mosaics of physical and hydraulic conditions sculptured by river sediments and flow are two primary determinants of where different species can and cannot live. A vital third determinant is water quality, comprising water chemistry, physico-chemistry (e.g. dissolved oxygen) and temperature. Water quality differs in rivers in different climatic and geological regions and, in any river, also changes along the system and through the year (Day & King 1995). Species will not occur in areas of suitable physical and hydraulic conditions, if the water quality is unsuitable. Small organisms in particular, such as aquatic invertebrates, are sensitive to water-quality conditions. Thus through their distribution patterns they give a clear indication of where and how this is changing (Eekhout *et al*. 1997). Fish may be halted in migrations along rivers by ‘barriers’ of reaches with unsuitable water quality, and may be unable to spawn or recruit young in areas previously suitable but where water temperatures have changed (King *et al*. 1998). Riparian and marginal vegetation may be less sensitive to water quality, but still exhibits coarse distribution patterns that appear to be linked to this (J. King, unpub. data).

Overall, species will only occur where all of these determinants – physical, hydraulic, chemical, physico-chemical and thermal – are suitable. Ecological determinants, such as competition, predation and biogeographical limitations, also influence distribution patterns, and so species may not occur where the primary abiotic determinants appear to be suitable.

Undisturbed systems are generally recognised as the most efficient users of scarce resources (Galat 2002). Studies of aquatic invertebrates in 30 Western Cape rivers in South Africa revealed that these undisturbed systems have both river and catchment signatures: each sample of invertebrates ‘knew’ which river and catchment it belonged to (King & Schael 2001). Research is continuing, in an attempt to reveal the nature of these signatures – perhaps they are caused by unique species in each river (biogeographical cause) or by unique combinations of species (possibly due to geochemical or other catchment characteristics). Whatever the cause, the evidence indicates that each river is slightly different, or maybe functioning in some slightly different way. This has as yet unknown implications for the management of these rivers.

Further, the same study revealed that disturbed rivers impact on the signatures in different ways. In MDS plots of invertebrate samples, samples with similar species plot close together and dissimilar ones further away. Increasing distance indicates increasing dissimilarity. In the MDS plot of undisturbed rivers, invertebrate samples from different rivers clustered by catchment, i.e. illustrated a catchment signature (King & Schael 2001). When the disturbed rivers were included, these were located in various positions on the plot in ways thought to reflect the severity of the disturbance on the ecosystem. Samples from rivers with mild disturbances were located well within their catchment cluster. Those from more disturbed rivers were located at the edges of their catchment cluster, whilst even more disturbance led to a group of rivers that were not included in established clusters. These rivers formed a new cluster in the centre of the others, possibly reflecting ‘generalised Western Cape rivers’ that had lost their identities, perhaps through the loss of their most sensitive species. The most disturbed rivers formed a group well outside all the other clusters, possibly reflecting the loss of even a Western Cape identity through loss of all but the most hardy, cosmopolitan species. The management implications are:

- until river and catchment signatures are better understood, the Precautionary Principle should be considered, as rivers in any one bioregion can no longer be assumed to be similar;
- disturbance affects river signatures and thus functioning of the river ecosystems, in ways which are as yet not understood.

In summary, growing understanding of the nature and functioning of rivers is leading to a recognition that their management should encompass all parts of the flow regime and all biotic and abiotic components of the ecosystem (King *et al.* in press). Rivers within any one bioregion may be less similar than previously thought, and disturbances to them may be affecting ecosystem nature and functioning in ways not yet understood. All of these factors should be taken into account in river rehabilitation projects.

2.2 Human impacts on river ecosystems

2.2.1 Alteration of rivers through human activities

Rivers were the arteries for the development of early civilizations, and in modern societies they remain central to local and global economies. Across the world, quality of life can be assessed in terms of the availability of fresh water, and most of this comes from rivers. In regions where rivers are located far from human settlements, there has been a trend to construct water-resource projects to bring the water from the rivers to the people (Hart 1992; Peace 1992). In most countries, particularly semi-arid or densely populated ones, river water has been stored in times of plenty to assure supply during drier periods. In countries well endowed with water, water-supply technology has lagged behind those of drier regions (Peace 1992), but in all countries, manipulations of one kind or another have gradually changed the distribution and nature of the world's surface waters.

Occupation of land by early civilizations was usually followed by land clearance for farming. Early water-supply schemes were constructed to provide water for agriculture and domestic use. Later, structures were developed to prevent flooding of crops and human settlements (Hart 1992; Teclaff & Teclaff 1973). In the 17th and 18th centuries rivers provided important transportation routes, and by the early 19th century, rapid industrial expansion, population increase and urban growth, had led to a dramatic increase in the development of industrial, municipal and rural water supplies and irrigated agriculture (Elliott *et al.* 1998; Petts 1988).

During the 20th century, modification of river flow became both widespread and intensive, through national dam-building programmes, inter-basin transfers of water, hydro-power schemes, direct abstraction of run-of-river flow, and catchment land-use practices that affected the movement of water into the river. Further modifications of rivers included channelization, canalization, their use as disposal systems for wastewater, sewage (O'Keeffe 1989; Hart 1992; Maddock 1999), and litter (e.g. cans, plastics, rubble, shopping bags, used car parts; (Armitage & Rooseboom 2000), their straightening and dredging for navigation purposes (Brookes & Hanbury 1990), and their use as fish farms (Brown *et al.* 1997).

2.2.2 Issues influencing river management

In the past, rivers were seen as hydrological systems: problems caused by them, or development of their water resources, were addressed with engineering approaches. Their management primarily focused on the consumptive needs of society, and failed to recognize that the donor rivers are dynamic, functioning ecosystems sculptured by their natural supply of water and bound to change as their water supply changes (Orsborn & Anderson 1986; Warner 1996; Carr *et al*. 1999). In water-resource developments, economic values tended to out-weigh moral and environmental values, discouraging the efficient use of water (Stanford *et al*. 1996), and there was little awareness that the valued attributes of rivers would change as their water-resources were developed. The situation was further exacerbated by the common practice of allocating the responsibilities of water delivery and catchment management to separate authorities, with actions by one often significantly impacting the outcomes desired by the other (Brizga *et al*. 1999a). Different land-use strategies by different landowners have further confounded coherent management of the whole resource, with an obvious need for overall management to be vested in catchment authorities. Most individual landowners do not have the resources to maintain the rivers on their properties, but nevertheless remain reluctant to hand over this task to a catchment authority (Beaumont 1981).

The low priority given in the past to maintaining healthy rivers has been reflected in the lack of environmental laws protecting them, and the under-pricing of water from them. This is changing as awareness has grown of the natural attributes and of the many resources rivers supply in addition to water. Rivers are resources shared by diverse groups of people, all of who have an interest in their exploitation. Nearly all countries depend on healthy, well-functioning rivers for a wide range of services, from flood attenuation to the reliable delivery of good-quality water. In addition, particularly in developing countries, millions of people depend on rivers for a range of goods, such as fish, building materials and medicines. These riparian subsistence users of rivers, usually among the poorest of people, are often the most impacted but least consulted about water developments that will affect their river and lifestyle (Ho 1996). There is also growing awareness, due of their continuing degradation, of the value of rivers as areas of recreation (water sports, fishing) and ecotourism (Schneiders *et al.* 1993; Shuman 1995; Ho 1996; Warner 1996; Maddock 1999).

2.2.3 The move toward environmentally responsible river management

Today, sustainable use of river systems is an increasingly important target for all countries. Governments are attempting to reconcile immediate socio-economic demands for water with long-term sustainability of the donor systems. Legitimate demands for water have been extended beyond the traditional ones of urban, industrial and agricultural use, to include those for maintenance of the river systems themselves, both in terms of consumptive water use and that needed for maintaining aquatic habitats (Stoffberg *et al*. 1994; Nilsson & Brittain 1996; South African Water Act of 1998). With the options for further cost-effective supply schemes now close to exhaustion, continued provision of good-quality water is more than ever dependent on environmentally sensitive river management.

Rehabilitation of rivers is thus becoming an expanding area of investment by public water management bodies in many countries. Rehabilitation marks a new era in the approach to river management (Regier *et al*. 1989; Sear 1994; Muhar *et al.* 1995), arising from increased awareness of the need for a healthy environment, and of the demand for accountability regarding environmental degradation. No longer seen merely as a constraint on development, the emerging management paradigm is based on ecosystem management and collaborative decision-making (Boon *et al.* 2000).

2.3 Concepts of rehabilitation

2.3.1 A paradigm shift in Ecology and Conservation Biology

Ecology and Conservation Biology are undergoing major conceptual development regarding the nature and functioning of ecosystems (Rogers & Bestbier 1997). Two opposing views have emerged that are influencing the field of river rehabilitation. These are: the traditional Balance of Nature paradigm and the new Flux of Nature paradigm.

The traditional focus of conservation has followed a Balance of Nature paradigm, whereby ecosystems are seen as static and, if not subject to disturbance, in equilibrium. Conservation programmes have focused on maintaining stable and homogeneous landscapes by managing desired species at chosen populations levels. This has been achieved by excluding anthropogenic influences and controlling populations of species in enclosed areas. The protection of desired ecosystems from all natural and anthropogenic influences, external to the conserved ecosystem, resulted in a general reduction in natural variation in the managed system, and a loss of ecosystem resilience (Rogers & Bestbier 1997). This dampening of ecosystem change maintained ecosystem functioning at a chosen level.

Proponents of the Balance of Nature view believe that humans exist outside of what is considered to be a natural ecosystem (Jackson *et al*. 1995), unless the humans are indigenous peoples or hunter-gatherers. In this context, human influences are undesirable, and a natural ecosystem is considered to be one that has not been impacted by modern society in any way.

The new Flux of Nature paradigm recognises that ecological systems are rarely in equilibrium, but are dynamic and heterogeneous in nature (Rogers & Bestbier 1997). Since ecosystems are continually changing, the biological constituents present today in an ecosystem will not necessarily be the same as those present in the future. Ecosystems are thus seen as being temporally unique, sculpted by isolated climatic events and biological invasions (Jackson *et al*. 1995).

This new perspective impacts on the way conservation authorities manage natural landscapes. The focus of conservation shifts from species conservation, through population control in closed stable ecosystems, to an ecosystem-level perspective where temporal and spatial heterogeneity is emphasised across all scales from genes to landscapes (Rogers & Bestbier 1997). The two new concepts introduced by the Flux of Nature paradigm are scale and heterogeneity.

Scales are addressed in hierarchy theory (Allen & Starr 1982; O'Neill *et al.* 1986; cited by Rogers & Bestbier 1997), and heterogeneity in the concepts of biodiversity (Section 1.3.3) and patch dynamics (White & Pickett 1985). Patch-dynamics theory recognises that landscapes are mosaics formed on geological and climatic templates, all hierarchically nested within each other and dictating environmental conditions at different scales. Each level of the hierarchy is constrained by the nature of the level within which it is nested. For example, riparian plants along the foothill zone of a river will not be fynbos species, unless the catchment itself is within the fynbos bioregion. On these templates, the mosaics are further shaped by the biological responses of organisms interacting with each other and their environment. For instance, foothill riparian plants may change from a reed community to a tree community with time, or vice versa, and then change back again if physical forces such as floods impinge. In the absence of human activity, these primary geological and climatic forces, and the biological responses, are the main driving forces creating the mosaics, with landscape patchiness changing through space and time as different controlling factors contribute to its appearance (Rogers & Bestbier 1997). Where human manipulations intervene, the mosaics may change in ways not likely to occur without such intervention.

The Flux of Nature paradigm recognises that anthropogenic influences on natural landscapes should not be treated as a problem: human intervention in the management of landscapes is seen as unavoidable. Any manipulation is a form of landscape engineering, regardless of whether it is designed to achieve some natural state or not. This engineering may be passive, where natural forces are allowed to operate, or active. The end result, even in National Parks, is a landscape defined by humans (Rogers & Bestbier 1997). All management is located somewhere on the continuum from active engineering to pure custodianship of ecosystems. Man is seen as an integral part of ecosystems, and the landscape is a manifestation of value judgements, that reflect a mix of anthropogenic and natural forces.

This contrasts with the traditional Balance of Nature view, where any human influence or impact on a natural ecosystem is seen as a disturbance with a negative influence (Jackson *et al*. 1995). Even though modern thinking considers human intervention necessary for conservation and management of natural landscapes, anthropogenic influences are generally seen as detrimental to natural ecosystems.

2.3.2 Disturbance of natural ecosystems

White & Pickett (1985) define disturbance in general terms as, "…any relatively discrete event in time that disrupts an ecosystem, community or population structure, and changes resources, substrate availability or the physical environment". Ecosystems, communities and populations are biological attributes, while resources, availability of substrata and the physical environment refer to variables constituting an organism's habitat. This definition is broad, encompassing environmental fluctuations and destructive events, whether these occur naturally or are human-induced.

By contrast, Resh *et al*. (1998, cited by Calow & Petts 1994), view disturbance as non-natural (human induced) and of a physical nature, thus excluding natural variability and cyclic heterogeneity of ecosystem functioning. Their definition of disturbance is "..those physical and biological events that occur outside the normal range of predicable frequencies, severities or intensities for that system".

After a disturbance, the values of the parameters defining the ecosystem in question change. The two paradigms concerning ecosystem functioning discussed in Section 1.3.1 differ in their interpretation of the consequences of these changes.

Proponents of the Balance of Nature view believe that outside (non-natural) forces should be managed to minimise the impacts they will have on river ecosystems. When the ecosystem is disturbed the aim would be gradual recovery to its pre-disturbance state through a process of natural succession. Succession is defined as "…the non-seasonal, directional continuous pattern of colonisation and extinction on a site by populations of plants and animals" (Begon *et al*. 1990). Whether or not natural succession is capable of returning a disturbed system to its pre-disturbance state will depend on the severity and persistence of the disturbance. Severely disturbed systems may require some form of intervention on the part of the river manager to aid recovery.

Those who propound the Flux of Nature paradigm believe that ecosystems are continually disturbed (both by natural and anthropogenic forces) and respond to differing levels and kinds of disturbance in various ways (Jackson *et al*. 1995). In this view the biological constituents present in an ecosystem can change with time. Such a continually changing ecosystem places no importance on the original natural (reference) condition, in contrast to the pre-disturbed reference state recognised under the Balance of Nature paradigm. In the Flux of Nature view there is no value in trying to return a disturbed ecosystem to its pre-disturbance state, since disturbance and biotic invasions are seen as normal and natural phenomena. It could be argued that if any anthropogenic impact is natural, then *rehabilitation* is not required. Or is it? Why might a near-natural ecosystem be of value?

2.3.3 The nature and value of naturally-functioning ecosystems

A naturally-functioning ecosystem is often considered to be one that is biologically diverse (Meier 1998), i.e. containing a large number of organisms for that kind of system. A common interpretation of this would be to view an area with large species diversity as one with high biodiversity (diversity of biological organisms). However biodiversity should take diversity of the whole hierarchical classification of living organisms into account, not just species diversity.

Different living organisms are commonly separated at the species level. Closely related species are grouped together into a common genus. Families encompassing a variety of similar genera are in turn grouped into orders. An area can be considered to have a high biodiversity if it supports a large number of different species, from many different genera, families and orders. Numbers alone, however, may give a misleading picture. Some river ecosystems may contain many different species of animals and plants, a large number of which are alien species. According to Karr (1996, cited by Meier 1998) such an area does not have high species diversity. This is implicit in Karr's definition of ecological integrity: "..the capacity to support and maintain a balanced, integrated, adaptive ecosystem, having the full range of elements (genes, species, assemblages) and processes expected in the natural habitat of a region". Thus Karr views an ecosystem disturbed by man through the introduction of alien species as having a lower ecological integrity and than one not so disturbed.

Rutherfurd *et al*. (2000) discuss what constitutes a "good or healthy" community. They suggest that a healthy river community is one that is pristine, again referring to a natural state uninfluenced by humans. They continue with a definition of a good community as one "containing a diversity of species, a significant proportion of which will be intolerant of 'bad' stream health (such as poor water quality, and reduced habitat diversity)". The presence of aquatic species that are sensitive to disturbance events is thus an indication of a relatively undisturbed and healthy stream. Such a system would also support the hardier species, and thus have a higher diversity than a disturbed system, where only the hardy species remain. Again, the emphasis is on the desirability of natural levels of species, not just a large number.

Meier (1998) and Rutherfurd *et al*. (2000) both discuss a 'healthy ecosystem' as one that supports a biologically diverse, indigenous community. In doing so they define ecosystem health in a biological sense, and do not specifically mention physical habitat. Required levels of species diversity and sensitive species will not be obtained, however, if appropriate physical habitats and chemical conditions are absent. The central premise of rehabilitation is that these underlying abiotic conditions dictate the kinds and diversity of biota that occur (Rutherfurd *et al*. 2000). Attempts to return ecosystems to their pre-disturbance states have centred on the reintroduction of appropriate habitat (physical, hydrological, chemical, and access to wetlands and floodplains), assuming this will stimulate the desired biological response (Rutherford *et al*. 2000).

The desired response may be an increase in the population numbers of selected species. Examples would be the regeneration of indigenous vegetation in an area previously occupied by alien species, or a general increase in the variety and abundance of aquatic organisms due to an improvement in water quality.

Physical habitat, although one of the primary variables affecting the distribution of populations and often the aspect most addressed in rehabilitation, is not necessarily the one most likely to influence them. Population size is controlled by a myriad of factors, the combination of which may be different for different species. Rutherfurd *et al*. (2000) discuss the concept of a limiting requirement, with this being defined as "...that resource which is essential to the stream community, but is most lacking...". As an example, if the distribution of a species in a stream reach is limited by an intolerably high level of pollution and a lack of suitable physical habitat, it would be fruitless to attempt to enhance its numbers by increasing the presence of suitable physical habitat without correcting the pollution levels. In this example, each variable becomes a limiting requirement if the other one has been met.

Even though the role of biodiversity in the functioning of ecosystems is currently under debate, with conflicting answers from experimental and the theoretical studies, some trends are emerging (Welcomme 2001). Community diversity of aquatic organisms has been stated as being important for the productivity, stability (Begon *et al.* 1990) and aesthetics of inland water ecosystems (Welcomme 2001). Diverse systems have been shown to be more productive than simpler ones, but only up to a certain level of diversity; beyond this the addition of new species may not increase productivity (Baskin 1995, cited by Welcomme 2001). This relationship has been questioned by Le´veque (1995, cited by Welcomme 2001) who suggested there is no simple relationship between biodiversity and ecological processes such as productivity because of the complexity of ecological processes that may influence productivity.

Theory on the relationship between diversity and stability is similarly contradictory. MacArthur (1955, cited by Welcomme 2001) and Elton (1958, cited by Welcome 2001) stated that complex ecosystems, that is, ones with high biodiversity, are more stable and better able to withstand disturbances. Evidence for this from natural systems was hard to produce, however, and was contradicted by May (1971, cited by Welcomme 2001) who stated that diversity led to instability. Whether or not diversity favours stability, the concept of stability is still important. Stability usually implies a measure of the resistance and the resilience of an ecosystem to a disturbance. Resilience describes the speed with which a community returns to its former state after it has changed from that state, and resistance describes the ability of the ecosystem to avoid displacement from its former state in the first place (Begon *et al*. 1990). Like productivity, stability of ecosystems has been shown to increase up to a point, and then it appears to become independent of diversity (Welcomme 2001). Species that may appear to be redundant, however, could have different tolerances to stress or other environmental changes and thus have a role to play if conditions change (Chapin *et al*. 1997, cited by Welcomme 2001).

There have been attempts to calculate a monetary value for the aesthetic appeal of an ecosystem that is biologically diverse. Loomis & White (1996, cited by Welcomme 2001) placed economic values on threatened species in the USA, by calculating the 'willingness to pay' to save a threatened species. Moyle & Moyle (1995, cited by Welcomme) reviewed several strategies for placing market value, ecosystem value, existence value and inter-generational value on endangered species in order to promote their conservation.

In conclusion, increased productivity and stability of ecosystems both appear to have at least tenuous links with increased biodiversity. Stable, productive ecosystems would appear to offer a lower maintenance and potentially more economically viable management option than unstable, poorly productive ecosystems. A reasonable rehabilitation option could then be to realise the potential of each ecosystem in these two regards, to the extent possible within the project.

2.4 Definition of terms used in rehabilitation

There is a wide range of terms that relate to river management. A comprehensive list is given by Petts *et al*. (2000). Four terms relate directly to the field of managing disturbed rivers: *restoration*, *rehabilitation*, *remediation* and *enhancement*. These terms, usually used loosely and encompassing a wide range of objectives, are defined below. Four further terms used in this field: preservation, limitation, mitigation and creation, are also discussed briefly.

2.4.1 Restoration

Restoration is the most common term used to describe rehabilitative management of freshwater ecosystems. There has been no consensus in the literature on what constitutes *restoration*, or how this differs from *rehabilitation*, with the two main views reflecting the two paradigms of natural ecosystems and of the extent to which *restoration* is thought to be achievable. Calow & Petts (1994), Jackson *et al*. (1995), Meier (1998), Fogg & Wells (1998) and Petts *et al*. (2000) follow a Flux of Nature view. Rutherfurd *et al*. (2000) follow a Balance of Nature view.

Meier (1998) defines *river restoration* as, "…an attempt to bring the river back to as high a level of ecological integrity as possible, taking into account the prevailing socio-economic, political, and technological constraints. In highly managed rivers, the objective should be to maintain a healthy ecosystem that is able to meet the societal needs in a sustainable manner." Meier acknowledges that restoring a river to natural levels of ecological integrity is difficult, and sometimes unattainable because of the length of time that the river has been modified, and the degree of modification of the channel. Calow & Petts (1994) agree that returning a river system to a pre-disturbance state is highly improbable. They argue that this is largely attributable to a lack of pre-disturbance data or accurate historical records to guide the process.

Fogg & Wells (1998) and Petts *et al*. (2000) both define *restoration* as: "…the process of returning an ecosystem as closely as possible to pre-disturbance conditions and functions." They do not believe ecosystems can be recreated to resemble their pre-disturbance conditions since these are continually changing anyway. They see the goal as creation of a self-sustaining ecosystem, which does not degrade further and therefore, requires no intervention to maintain the 'restored' state. They have used 'restoration' to describe any level of intervention directed toward a "former natural condition".

Thus, Calow & Petts (1994), Fogg & Wells (1998), Meier (1998), Jackson *et al*. (1995) and Petts *et al*. (2000) all see *full restoration* as unlikely to succeed, either because of a dearth of historical records to guide the process, or because the ecosystem has been largely modified over a long period of time. As a result they use the term *restoration* loosely, to cover a wide range of management options where the objective is some degree of return of a degraded ecosystem toward its prior state. Following their Flux of Nature view; management of an ecosystem to maintain its restored state, or sustainable human use of the restored river ecosystem, can be encompassed within *restoration* activities. Thus, they view restoration as an umbrella term encompassing all the different levels of intervention, such as rehabilitation, mitigation and enhancement.

According to the Balance of Nature view, human impacts on ecosystems are a non-natural disturbance. Systems used on a sustainable basis by humans cannot be restored, because the impacts associated with that use, change the ecosystem from its pre-disturbance state. Thus, Rutherfurd *et al*. (2000) define a *restoration* project as one that achieves five objectives. These are:

- restore the natural range of water quality;
- restore the natural sediment and flow regimes;
- restore a natural channel geometry and stability;
- restore the natural riparian plant community;
- restore native aquatic plants and animals.

The aim of *restoration* according to Rutherfurd *et al*. (2000) is to return the degraded or impacted ecosystem to its pre-European condition. This is taken to mean the historically natural state; that is, the state prior to intensive human settlement. They recognise a further difficulty in attaining this pre-disturbance state, over and above those given by Meier (1998) and Fogg & Wells (1998). This is the difficulty in isolating a section of the river from impacts that occur either upstream or downstream. In their view, *river restoration* is only achievable if, "…the entire stream network, and most of the catchment surface, are also restored".

Rutherfurd *et al*. (2000) give an example of what they consider to be a candidate for *restoration*. The Thurra River in East Gippsland, Victoria, Australia, has a catchment area of 350 km^2 . The riverine corridor is entirely intact except for a 3 km stretch in the upper reaches where some grazing occurs on one bank, as well as logging which occurs outside of the riparian zone. If livestock access to this grazed bank was prevented, and the riparian zone was allowed to regenerate naturally, the river corridor could be considered to be restored naturally as there would be no threats from upstream or downstream and the in-channel condition is good. No mention was made of the role native riparian and aquatic species should play in assessing the success of such a restoration effort.

2.4.2 Rehabilitation

The term *rehabilitation* is defined less clearly and discussed less widely than *restoration*. There appear to be two main definitions, reflecting the Flux of Nature and Balance of Nature paradigms.

The Flux of Nature view is given by Fogg & Wells (1998), who define *rehabilitation* as, "…making the land useful again after a disturbance." The aim here is not to return the ecosystem to a pre-disturbance state but to achieve a stable landscape that supports the natural ecosystem mosaic. The term *rehabilitation* is used to describe stabilisation of a degraded river, with due regard for the various land-use activities already present in the catchment. The emphasis by Fogg & Wells, again, is to achieve an ecosystem that can be used on a sustainable basis by humans. Petts *et al*. (2000) use a similar approach, defining *rehabilitation* as, "…a partial return to a pre-disturbance condition, usually linked to fish or wildlife habitat. To return degraded habitats to a pre-existing condition (e.g. dredging backwaters that have filled with sediment; forming riffles, or changing the plan form of channelised reaches; or planting riparian buffer strips)." Again the emphasis is on achieving some greater degree of naturalness whilst recognising human demands on the river. Confusingly, *rehabilitation* of this kind requires knowledge of the historical condition, as does their definition of *restoration*.

Rehabilitation has also been used within the Balance of Nature paradigm, to describe the returning of an impacted ecosystem toward its pre-disturbance state, with the recognition that neither the entire stream network nor the greater part of the catchment can be rehabilitated. Rutherfurd *et al*. (2000) allude to this concept, although they give no clear definition of *rehabilitation*. Their aim of 'rehabilitation' is to "…create a stream that, although only resembling the pre-European condition, is nevertheless an improvement on the degraded stream". If the pre-European condition is attainable *restoration* could be achieved. Where this condition is not attainable, Rutherfurd *et al*. (2000) offer 'remediation' as a suitable alternative, and provide an example of the difference between *restoration*, *rehabilitation* and *remediation*.

Urban streams in Melbourne, Australia, were originally low-energy rivers consisting of pools separated at low flow by grassy chutes, commonly described as chains of ponds. Most such streams in urban areas are now channelised and incised, with larger flood peaks during storm events, and a greater than natural stream power. They cannot be *restored*, due to hardening of the surrounding catchment causing changes to their water chemistry and patterns of flow. Reversing the hydrological and chemical changes would require removing or negating the influence of the urban developments on the streams. As large-scale reverses in catchment land-use are probably impossible, *rehabilitation* could have the goal of returning the channelised streams to chains of ponds within the framework of the surrounding urban catchment. Again, this is probably not viable because of the increased flood peaks and water-quality problems generated by the catchment. Thus, the only option might be some kind of *remediation* or *enhancement*.

2.4.3 Remediation and enhancement

Rutherford *et al*. (2000) define the aim of *remediation* as "…to improve the ecological condition of the stream, but the end point of that improvement will not necessarily resemble the original state of the stream". This is similar to the definition of *enhancement* given by Calow & Petts (1994), and Petts *et al*. (2000): "…improving the current state of an ecosystem without reference to its initial state." Remediation or enhancement may be attempted in order to mitigate the effect of a disturbance or a group of disturbances and to provide optimal conditions for a highly valued species such as a game fish (Petts *et al*. 2000).

Improving the ecological condition of a stream may be done in a number of ways, depending on the objective to be achieved. Approaches may be applied singly or in combination. Some examples include:

- stabilising unstable river banks, by planting indigenous riparian vegetation;
- filtering agricultural runoff before it enters rivers, to remove pesticides and fertilisers;
- filtering polluted water, and installing measures to buffer the introduction of further pollutants into a river;
- clearing alien vegetation and replanting indigenous vegetation from river banks;
- creating artificial riffles for fish to spawn, in rivers where no riffles existed previously.

Remediation is distinguished from *restoration* and *rehabilitation* by the lack of a goal linked to the pre-disturbance state. In the above example of the Melbourne urban streams, a *remediation* or *enhancement* solution could be to create artificial riffles, which do not naturally occur in them. This would improve the streams by oxygenating the water and increasing the diversity of the in-stream habitat (Rutherfurd *et al*. 2000).

In summary, r*estoration, rehabilitation, remediation* and *enhancement* all aim to improve the level of ecological functioning of a degraded river ecosystem. They differ in the degree to which they aim to return the degraded system to its pre-disturbance state. In the next section, management options are defined that, although related to water management, do not have the aim to actively intervene within the boundaries of the river channel.

2.4.4 Other terms relating to water management.

The following terms illustrate the further range of options available to river managers. They are too vague to be of specific use for rehabilitation projects.

Petts *et al.* (2000) define *preservation* as "…the maintenance of functions and characteristics of an ecosystem in its desired state…not requiring rehabilitation". This implies protection in a static sense and restriction of a change in that state (Petersen 1991), with active management to sustain the 'desired state'. Thus, the ecosystem is not self-sustaining. Petersen (1991) suggested that rare species may be preserved in zoos or in protected habitats, but nature (taken to mean communities or ecosystems) in a broader sense should be conserved. He defined *conservation* as "...the planned management of a natural resource to prevent exploitation, destruction or neglect". The term implies management in a dynamic sense with less rigid control on ecosystem structure.

Petts *et al.* (2000) additionally define:

- *limitation* as "…actions to limit catchment development, often involving land-use and conservation planning";
- *mitigation* as "…actions to avoid, to reduce, or to compensate for, the effects of environmental damage";
- *creation* as "…bringing into being an ecosystem that previously did not exist on the site".

2.5 Summary

There is no consensus internationally about what constitutes *river rehabilitation* or *river restoration*. The two terms are used loosely to describe a variety of projects with different goals. The ambiguity is largely a result of the value that different people place on the natural environment and what constitutes a natural environment. Proponents of the Balance of Nature paradigm believe restored river systems should be protected from outside (non-natural) human forces that change ecosystems, whilst those of the new Flux of Nature paradigm view ecosystems as continually disturbed, both by natural and anthropogenic forces, and responding to differing levels and different kinds of disturbance. In their view, the biological constituents present in an ecosystem will not necessarily be the same as those in the future. Whichever view is held, there is a common desire to return a degraded river ecosystem to a more natural state.

The underlying assumption behind all river rehabilitation projects has been that the re-introduction of a variety of physical habitats will improve the ecological functioning of the ecosystem being rehabilitated.

3. DEGRADATION: PHYSICAL CHANGE

3.1 Introduction

Physical disturbances affecting river systems may be classified into two groups: those occurring outside the macro-channel of the river in the catchment, and those that impact on and within the macro-channel. The macro-channel is defined as the wider river channel that encompasses the active (usually wet) channel and all bankside areas, floodplains, secondary channels and flood terraces, to the furthest extent reached by low-recurrence interval floods.

Both kinds of disturbance, and their effects on the river, are best understood within the context of the processes that form the catchment topography and the factors that affect the rate at which these operate. This is discussed in Section 3.2, with disturbances that affect the river's catchment addressed in Section 3.3 and disturbances within the macro-channel in Section 3.4.

3.2 Sediment processes at the catchment scale

Catchment hillslopes occupy most of the land surface, with the exception of terraces and floodplains formed by river deposits (Selby 1993). Hillslopes are formed by the weathering and removal of rock and soil by a number of processes that operate at different spatial and temporal scales. The three dominant processes are mass wasting, soil erosion and solution (Selby 1993). *Mass wasting* is the 'downslope movement of soil or rock material under the influence of gravity without the direct aid of other media such as water, air or ice' (Selby 1993). Mass wasting includes landslides, and is an important process in the development of hillslopes, especially in steep and mountainous regions. *Solution* occurs when rainfall percolates among rocks and through the soil. Such water movements cannot carry coarse material, but do dissolve finer material from the substrata that are then removed in the flowing water. If water at the soil surface cannot percolate downwards, it will move across the surface. Hence, *soil erosion*, or the removal of surface material by water, is most active when *solution* is least active (Kirby 1980). *Soil erosion by water* is part of a greater process of erosion that also includes detachment and removal of soil and rock by wind, waves, flowing ice and mass movement (Selby 1993).

The three processes operate episodically and are dependant upon the availability of an energy source and a medium to transport the eroded material. The ultimate energy source, which directly controls weathering processes, is solar radiation. Indirectly, solar radiation also controls the movement of water through the hydrological cycle.

Mass wasting is an important process with serious economic and social implications, since it tends to widen valleys and lower slope gradients (Kirkby 1980). In terms of anthropogenically-induced disturbance to catchments, however, neither mass wasting nor solution are viewed as of primary relevance, whilst soil erosion is, through its link with water movement. Thus, only soil erosion is addressed in this Chapter, with the hydrological processes relevant to soil erosion addressed in Section 3.2.1, and the factors that control soil erosion in Section 3.2.2.

3.2.1 Hydrological processes related to soil erosion

Soil erosion may be seen as a problem when the rate at which it occurs is accelerated beyond what is considered to be normal for a particular hillslope. According to Kirkby (1980) there are three avenues of research on soil erosion. The first considers the significance of soil erosion rates through geological time. The second addresses the immediate controls on soil erosion rates and the processes affecting those controls. The third assesses the spatial and temporal variation of soil erosion rates. The following discussion focuses on the second view, since this is the time and space scale at which human disturbance occurs.

Rainfall and hillslope hydrology: infiltration

Water is the most common agent by which soil on hillslopes is moved (Selby 1993). The hydrological cycle describes the movement of water from its source, in the form of precipitation, its movement through the landscape and eventual return to the atmosphere.

Infiltration is that part of the cycle where water enters the soil (Baird 1997). The most obvious controls on the rate of infiltration are the antecedent soil-moisture conditions and the physical properties of the soil. The finer the texture of the soil and the less ordered the physical structure, the lower the infiltration rate (Baird 1997). Other properties of soils that affect the rate of infiltration (Selby 1993) are:

- the amount of water available at the soil surface, since water will only enter the soil body if there is a film of water present on the soil surface;
- the nature of the soil surface - some soils are hydrophobic and resist wetting of their surfaces;
- the ability of the soil to conduct water away from the surface - saturated soils may not be able to absorb more water vertically downwards.

The vegetation cover on a hillslope also affects the rate of infiltration, as well as affecting soil moisture directly (Baird 1997). The chemical nature of the soil and the level of soil moisture are altered through nutrient and water uptake by plants. Root growth also forms pathways for the conduction of water through the soil surface. This is thought to increase the overall hydraulic conductivity of soil immediately around the plant base (Baird 1997), with infiltration rates beneath vegetation being greater than on non-vegetated slopes.

Rainfall and hillslope hydrology: interception

Vegetation intercepts rain on its path to the soil surface. Rain hitting the leaves may collect on them and drip off onto the soil surface, or it may run down the trunks of the plants as stem flow (Selby 1993). The amount of water lost to the hillslopes because of interception depends upon the type of vegetation: overall losses are generally greater from evergreen canopies than from deciduous ones. Air temperature also influences interception loss, with higher rates of evaporation at higher temperatures. The duration, amount, intensity and frequency of rainfall all affect the amount of pre-existing water on the plant, and thus the amount of water that can be intercepted at any given time (Selby 1993).

Stem flow is important in the process of infiltration. By intercepting rainfall and so reducing the number of drops of rain hitting the soil surface directly, plants help to protect the soil from sealing. Sealing is caused by the physical and chemical breakdown of soil aggregates by rain-splash and the washing of fines into larger pores (Baird 1997). Seals block the soil surface during rainfall events, while crusts form during dry phases. Both decrease the infiltration capacity of the soil. Through contributions from stem flow (and in protecting the soil from sealing), plants help to increase the efficiency of infiltration and decrease the generation of overland flow.

The role of vegetation and its impact via raindrop erosion, as well as its influence on other processes of hillslope erosion, is further discussed in Section 3.2.2.

Rainfall and hillslope hydrology: runoff

Rain falling onto hillslopes generates overland flow, or runoff, in two basic ways. If the amount of rain arriving at the surface of the soil exceeds the rate at which water can infiltrate the soil surface, water is stored in small depressions on the hillslope until the depressions are overtopped. Overtopping causes the water to flow down the hillslope. This is called *infiltration excess overland flow* and is common in semi-arid and arid regions (Baird 1997). Conversely, when water is prevented from infiltrating water-saturated soils in topographic lows and areas adjacent to stream channels by a rise in the water table, excess groundwater and rainwater than contribute to overland flow once the surface-detention capacity has been exceeded (Baird 1997). This is termed *saturation excess overland flow*.

3.2.2 Factors controlling the rate of erosion

Climate and the geological nature of the catchment are the two over-arching influences on erosion (Selby 1993), directly controlling the rate at which soil may be eroded from a hillslope. The underlying geological structures determine the shape of the landscape, and the character of the soil, which in turn dictate the rate at which soil may be eroded and the type of vegetation that may occur on the slope. Vegetation type is also determined by, among other things, the climate (rainfall, temperature, hours of sunlight) of the region. Vegetation in turn influences soil structure and composition through root action, uptake of nutrients, provision of organic matter and protection of the soil from erosion (Selby 1993). These influences and others all operate together, and are expressed by the Universal Soil Loss Equation (USLE) described by Selby (1993).

USLE describes soil loss as a function of rainfall, soil erodibility, slope length, slope gradient, cropping management and erosion control factors. The equation was developed for use in soil-erosion studies in pasturelands with *infiltration excess overland flow* and has not been applied widely to uncultivated slopes with grass or tree cover. It does not apply to soils being eroded by mass wasting. The USLE has limitations because it was devised for conservation of soils in agricultural fields in the United States, but is useful because it provides insights into the main factors influencing erosion processes, namely, climate and raindrop erosion, soil, topography and vegetation.

Climate and raindrop erosion

Precipitation is considered the most important climatic factor influencing runoff and erosion (Selby 1993). Wind and temperature also exert some influence on runoff and erosion processes. In particular, temperature influences the type of precipitation, and can also change the adsorptive properties of the soil by freezing the surface of the soil, and can increase subsequent erosion through the breakdown of rocks in the freeze-thaw cycle.

A great deal of sediment that is transported down slopes was detached by raindrop erosion. Baird (1997) gives a good account of how raindrops dislodge soil particles from the ground surface, which are then transported downhill in the drops or in general overland flow. Raindrop erosion is influenced not only by slope, but also by vegetation cover, the texture of the surface substrata and the depth of the overland-flowing water. Baird (1997) summarises the effect of these three factors as follows:

- Vegetation intercepts rainfall and decreases the number of raindrops directly hitting the soil surface. Raindrops that collect on the leaf surface and then drop to the soil are sometimes larger than those hitting the leaves and so may increase the erosive force exerted by the rain on the soil surface.
- The texture of the soil surface affects the soil's cohesivity. Finer soils tend to have stronger forces of cohesion (forces of attraction between adjacent particles) that will resist the erosive forces of the raindrops more effectively than a coarser soil.
- Once a film of water has formed over the soil surface, the effectiveness of detachment and transport of sediment particles under raindrop impact depends on the depth of the water. Raindrop-induced turbulence is greater in shallow than deep water, and the erosive potential and transport capacity of water increases with increasing turbulence.

Topography and slope

Erosion across hillslopes is accelerated on steeper slopes. Most research on soil erosion has compared the rates on steep and gentle slopes. Steeper slopes are more likely to have a higher velocity of runoff (Selby 1993), which increases the potential for soil particles to be detached from the substratum. The ease with which soil particles are detached (the erodibility of the soil) from the hillslope depends upon the properties of the soil (Quanash 1985). The length of the slope affects the total amount of soil that can be removed by overland flow (Selby 1993), with all else being equal, long slopes usually provide more soil than short ones.

Vegetation

Hillslope vegetation exerts a control over the nature and operation of hillslope processes, by adding an element of roughness to hillslopes, by intercepting rain (Thornes 1990), and by affecting the rate of infiltration. The roughness provided by vegetal cover decreases runoff velocities on hillslopes through friction, and prevents channelling of runoff (Selby 1993) by deflecting flow in different directions as it moves downhill. Both slow down erosive processes.

Spatial and temporal variability in catchment erosion

The local controls on soil erosion of topography, vegetation cover and soil properties vary across the catchment (Thomas 1990). This variability results in different parts of the catchment responding differently to the same intensity and frequency of rainfall (Selby 1993). Land-use changes will also alter the erodibility of these different areas. By comparison with the high runoff generated from an eroded catchment, a well-vegetated catchment with a more permeable soil will experience a higher rate of infiltration, lower surface runoff and less surface erosion (Selby 1993).

Vegetation cover is not static but changes seasonally, and so the susceptibility of a hillslope to erosion by overland flow may vary temporally (Thornes 1990). Although the separate influences of rainfall, slope length and gradient, soil cropping and vegetation cover are recognised, the relative importance of each on a particular hillslope remains difficult to determine (Kirby 1980).

The sediment-production and erosion processes outlined above are all natural. It is evident from Selby (1993) and Thornes (1990) that vegetation plays an important role in the reduction of erosion on hillslopes. Erosion will thus be increased, with more sediment mobilised, if the vegetation is disturbed (Rowntreee 2000), which can happen through a variety of human activities.

3.3 Impacts of hillslope erosion on the river channel

The shape of a river's channel is a function of the climate to which it is exposed, the geological composition of the catchment that it drains (Gordon *et al*. 1992), and the fluvial processes at work within the channel. Channel dimensions change over time to accommodate the amount of water and sediment entering from the catchment. Streams are either supply limited or capacity limited depending on whether or not the ability of their flows to carry sediment exceeds the amount available (Gordon *et al*. 1992). A river with limited transport capacity is one that has flows unable to transport all of the material that is delivered to it. In such a system, sediments will be deposited, and the channel will aggrade. If the river's flows have the capacity to transport more sediments than are delivered to the channel, then degradation will occur as erosion of the bed and banks takes place. Both kinds of channels can be affected by increased or decreased mobilisation of catchment soils, and some may move from being degrading to aggrading systems or vice versa (Jewitt *et al*. 1998).

3.4 Disturbances to the river channel

Disturbances to the river channel usually involve re-shaping the bed or banks, or straightening, widening, deepening or narrowing the channel in other ways. These disturbances affect the sediment and flow regimes of the channel as detailed below.

3.4.1 Channelisation

The creation of new channels, or modification of existing river channels through straightening, narrowing, deepening, widening, or construction of embankments, can collectively be defined as channelisation. Alternative descriptive names include 'river regulation works' or 'river training works'. Other channelisation procedures, such as dredging, cutting, and removal of in-stream or bank obstructions, are sometimes classified as river maintenance (Gregory *et al.* 1985). Channelisation can be for flood control, for land drainage to create and maintain agricultural or urban land, for navigation purposes, or for reducing or preventing erosion of banks and the riverbed (Table 3.1). Channelisation for flood control or land drainage facilitates the rapid drainage of excess water to the sea (Andersen & Svendsen 1997).

Table 3.1 Case studies of channelised rivers

River and country	**Purpose of channelisation**	**Source**
American rivers	Flood control, land drainage	Keller 1975; Harvey & Watson 1986; Wissmar & Beschta 1998
Cape Town rivers, South Africa	Flood control	Brown 1992
Danish rivers	Land drainage	Brookes 1987; Iversen *et al.* 1993
Danube River, Austria	Flood control	Tockner *et al.*, 1998
Harper's Brook, U.K.	Flood control	Ebrahimnezhad & Harper 1997
Hout Bay River, South Africa	Flood control, land drainage	Beaumont 1981
Kissimmee River, U.S.A.	Flood control	Dahm *et al.* 1995; Toth *et al.* 1998
Kuils River, South Africa	Flood control	Wiseman & Simpson 1989
Latrobe River, Victoria, Australia	Flood control	Reinfelds *et al.* 1995
Laural Creek, Canada	Flood control, land drainage	Winter & Duthie 1998
Martin Dale Creek and Martin Dale Tributary, Mississippi, U.S.A.	Flood control, land drainage	Shields *et al.* 1997
Mississippi River, U.S.A.	Flood control, land drainage	Shields *et al.* 1995a
Missouri River, U.S.A.	Navigation	Harberg *et al.* (undated)
Morava River, Czech Republic	Land drainage	Sterba *et al.* 1997
Raisin River, Ontario, Canada	Land drainage	Watelet & Johnson 1999
Ravensbourne River, U.K.	Flood control	Tapsell 1995
Rhône River, Geneva, Switzerland	Flood control	Henry *et al.* 1995
River Alt, U.K.	Flood control, land drainage	Nolan & Guthrie 1998
River Brede, Denmark	Flood control, land drainage	Nielsen 1996; Vivash *et al.* 1998
River Cole, U.K.	Directing of water, flood control and land drainage	Nielsen 1996; Vivash *et al.* 1998
River Elbæk, Denmark	Flood control	Brookes 1990
River Idle, U.K.	Flood control, land drainage	Downs & Thorne 1998
River Lambourn, U.K.	Flood control	Brookes 1990
River Livojoki, Finland	Log transport	Tikkanen *et al.* 1994
Rivers Rhine and Meuse, The Netherlands	Navigation	Schropp & Bakker 1998; Van Dijk *et al.* 1995
River Skerne, U.K.	Flood control	Eden *et al.*, 1999; Nielsen 1996; Vivash *et al.* 1998
River Skjern, Denmark	Flood control and land drainage	Andersen & Svendsen 1997
Rivers in agricultural areas, Finland, Sweden, Denmark, U.S.A.	Land drainage	Petersen *et al.* 1992
Wraysbury River, UK	Flood control	Kondolf 1996
Whittle Brook, U.K.	Flood control	Nolan & Guthrie 1998

Channelisation of the river channel changes its hydraulic geometry and morphology. It creates a more uniform channel shape, frequently with a trapezoidal cross-section (Gregory *et al.* 1985; Newbury & Gaboury 1993a; Kondolf 1996; Downs & Thorne 1998; Nolan & Guthrie 1998; Eden *et al.* 1999; Maddock 1999).

3.4.2 Types of channelisation

Straightening

When a meandering river is straightened, the channel path is shortened, friction at bends is reduced, and the channel gradient (slope) is increased (Brookes 1987; Brookes & Gregory 1988; Luger 1998). Straightening may also involve confining braided systems to a single channel (Wissmar & Beschta 1998). The net effect of straightening is an increase in current speed under both high and low flow conditions (Henderson 1986; Luger 1998). As a result the channel will tend to become deeper and wider (Gregory *et al.* 1985; Henderson 1986).

Straightening affects the sediment and flow regimes of the river. The increase in channel slope will result in an increase in flow velocities in the straightened reach, which will increase the erosive capacity of the water and enable the transport of more sediment than is supplied by the natural channel from upstream. Any imbalance between sediment supply from upstream and increased erosive capacity will result in erosion of the bed of alluvial channels (Richards 1982). As the river incises, the gradient of the riverbed between the channelised and upstream reaches increases. This will result in increased erosion upstream of the channelised reach (Richards 1982; Brookes & Gregory 1988; Simon 1989), thus generating a knickpoint of erosion that will migrate upstream. Deposition of the eroded material will occur downstream (Richards 1982).

Straightening of the Obion River, USA resulted in the bed upstream of the straightened reach becoming incised to a depth of 6 m over a period of 10-15 years (Simon 1989). Similarly, on the Bunyip River in Australia, straightening resulted in the upstream migration of a zone of bed incision for many kilometres along the main river and its tributaries, eroding both the bed and bank and causing roads to collapse (Shields *et al.* 1995a; Brizga *et al.* 1999b).

Simon (1989) and Shields *et al.* (1995a) described the morphological nature of incising channels in a conceptual model of channel evolution that recognizes five or six stages of channel response to straightening (Figure 3.1).

- *Stage 1: pre-modified*. Characterised by stable, low-gradient and well-vegetated banks as a result of natural fluvial processes and land-use practices. Bank failure by mass wasting occurs infrequently.
- *Stage 2: construction (straightening)*. Existing banks may be reshaped or the entire channel repositioned during the construction of the new channel. In either case, the banks generally become steeper, higher and straighter. Channel width is increased, with vegetation removed to increase channel conveyance. The channel generally has a trapezoidal shape.
- *Stage 3: degradation*. This is characterised by rapid erosion of the channel bed (incision) due to the shortened channel path and increased current velocities, which result in an increase in bank heights. Bank steepening occurs when moderate flows attack basal surfaces and remove

toe material. Stage 3 is probably the most important one, determining the magnitude of subsequent channel widening, as the amount of incision partly controls the bank-failure threshold.

- *Stage 4: threshold*. Continued basal erosion heightens and steepens the banks. Bank failure occurs if the critical bank height is exceeded. Bank retreat and failure continue to develop a vertical bank face and an upper bank of 25-50 degrees. Moderate to high flows generally remove the material deposited at the toe. This retains the over-heightened and over-steepened bank profile, and gives the banks an eroded appearance.
- *Stage 5: aggradation*. Aggradation of the channel bed begins, often in the form of sand deposits on banks. Previously deposited material on the upper bank also moves down the slopes where it may be re-worked and deposited by river flows. The material deposited on the banks and bed develops to become a bench or small floodplain.
- *Stage 6: re-stabilisation*. There is a significant reduction of bank heights, due to channel aggradation on the channel bed and fluvial deposition on the upper bank and slough-line surfaces. Mass-wasting processes are less obvious than in stages 3 to 4 because bank heights no longer exceed critical heights. Woody vegetation extends up the slope towards the base of the vertical face and the former floodplain surface becomes a terrace. Bars gradually form, become vegetated and a meandering plan form is regained (Simon 1989; Shields *et al.* 1995a).

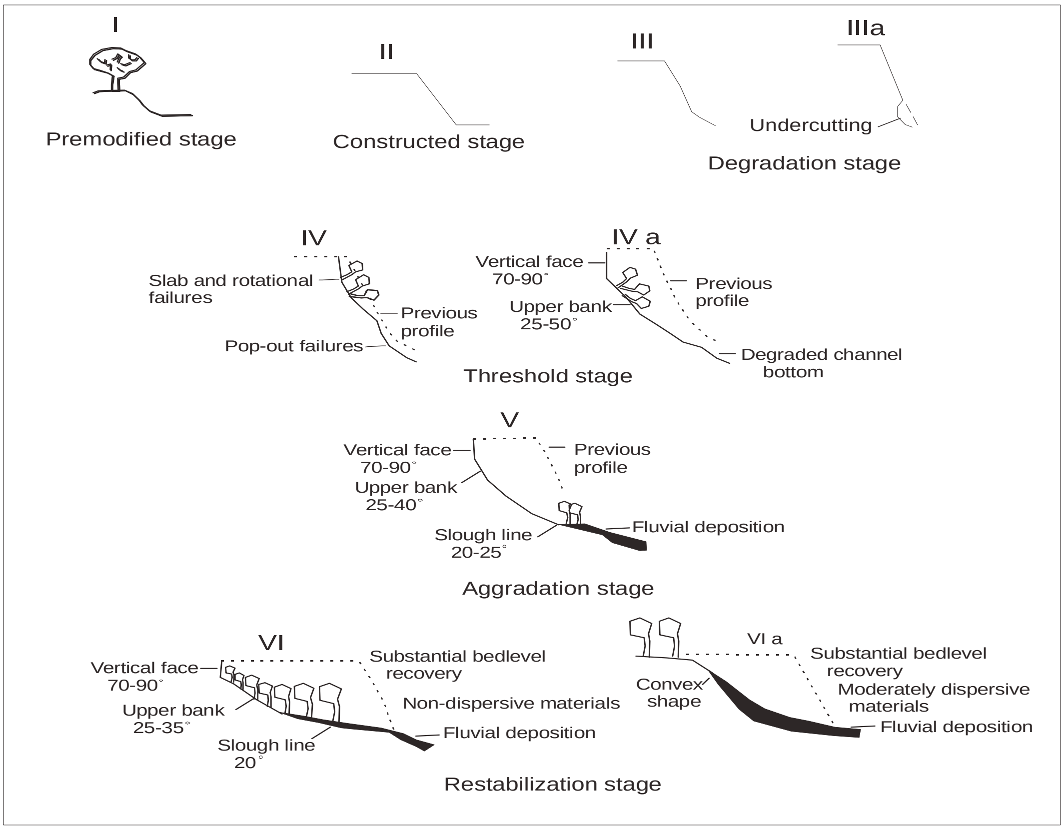

Figure 3.1 Stages of bank retreat and slope development (From Simon 1989, p 20)

Embankments

Creation of embankments (also called dykes or levees) along the river channel is often part of the straightening process. They are in general use in European countries such as the Netherlands and Denmark (Admiraal *et al.* 1993; Henry *et. al.* 1995; Andersen & Svendsen 1997). Embankments are used to restrict the flow in a central channel, to prevent the lateral migration of rivers and to separate the river from its original floodplain, thereby increasing the area available for development. Restricting water to the main channel prevents the overtopping of banks onto the developed areas. However, this limits the natural dissipation of energy that would occur during overtopping. Water with increased energy is then transferred to downstream reaches causing degradation and sedimentation further downstream (Maitland & Morgan 1997). Restriction of water between the embankments leads to increased current speeds in the channel and can result in incision of the riverbed. For example, after river straightening and embankment building, the bed of the River Rhine dropped by 7 m with the incision occurring up to 300 km upstream of the engineering works (Pearce 1993). The isolation of the river from its floodplain also prevents movement of silt from floodplain to river, decreasing sediment supply (Pearce 1993; Maitland & Morgan 1997). The Rhône River in Geneva, Switzerland used to be braided, until the construction of embankments along the river resulted in some channels being isolated from the main channel. The blocked channels only received seepage water through the embankments during low flows and river water during high flows. The embankments prevented the creation of new channels by impeding the lateral movement of the river, and a single channel system resulted (Henry *et al.* 1995).

Deepening

Deepening the riverbed is another way of increasing the capacity of the channel to carry floodwaters, and also allows the easier passage of boats (Brookes & Hanbury 1990). Deepening affects the sediment and flow regimes of the river because deepened reaches can serve as sediment traps affecting sediment supply to downstream reaches. Sediment deposition within the deepened reaches results in the reduction of flow depths and therefore a decrease in the capacity of the channel to carry floodwaters in the long term, although the flood carrying capacity will still exceed that of the original river (Gregory *et al.* 1985; Brookes & Gregory 1988; Brookes 1992). As an example, the deepened channel in the San Lorenzo River, U.S.A. was designed to carry a flood with a return period of 100 years, but sedimentation reduced the capacity to the magnitude of a 25-30-year flood (Brookes & Gregory 1988). Importantly, the trapping of sediment within the deepened reach is likely to result in reduced sediment loads further downstream with consequent possible problems of bed degradation (Simon 1989; Wyźga 1996; Ebrahimnezhad & Harper 1997).

Deepening of the channel will lower the water levels both in the channel and then in the groundwater adjacent to the channel. This, in turn, may affect the low flow conditions in the river during dry periods, because there is less groundwater to seep into the channel (Admiraal *et al*. 1993; Shields *et al.* 1997; Sterba *et al.* 1997). Channel deepening can reduce the frequency with which rivers overtop their banks, which is likely to reduce groundwater recharge and further reduce base flow (Gore & Shields 1995; Schropp & Bakker 1998; Wissmar & Beschta 1998).

Widening

Widening is another method of increasing channel capacity. It results in a decrease in current speeds due to the enlarged cross-sectional area of the channel, and hence a reduction in stream power per unit bed area. The reduced capacity to carry sediments may lead to sedimentation problems. For instance, banks may collapse if they are not stabilised after widening. Also, flow velocities will increase downstream if the downstream reaches are narrower than the widened reaches upstream, possibly resulting in increased bed erosion. Channel capacity will decrease again with time as low flows, which predominate for most of the time, deposit sediment on the channel margins. These may become stabilized with vegetation, to form permanent morphological features. Thus, there will be a tendency for the channel to revert to its original shape. The River Tame near Birmingham was widened for flood control in 1930. In the absence of maintenance, it reverted toward its natural width in less than 30 years (Brookes 1992).

Narrowing

Restriction of flow to a narrower channel will decrease the channel capacity of the river, and result in increased current speeds and local scouring of the bed. Secondary deepening may then occur, increasing the channel capacity again. Localised bed scouring usually results in increased sedimentation in downstream reaches (Admiraal *et al*. 1993; Pearce 1993; Wyźga 1996).

Maintenance

Brookes & Gregory (1988) group a number of activities such as weed control, removal of bank vegetation, dredging of accumulated sediments and clearance of rubbish from the channel as river maintenance. Urban rivers within the Western Cape have channels choked with water plants such water hyacinth (*Eichhornia crassipes)* and bulrush (*Typha capensis*), which are removed mechanically and manually in ongoing maintenance programmes (Jordaan, Oostenberg Municipality, pers. comm., 2000). Physical removal of other in-stream features is also common practice to enhance efficient conduction of water out of farming areas. In Australia, this usually takes the form of manual removal of debris, litter and larger accumulations of wood and other organic matter. (Rutherfurd *et al*. 2000).

Ecological impacts of channelisation

Channelisation impacts the flow and sediment regimes of rivers. As a result, some hydraulic biotopes may be lost (King & Schael 2001). This may lead to a decrease in species diversity, since many species require a specific flow type and/or substratum conditions and some are highly intolerant of change (Petersen *et al.* 1992; Admiraal *et al.* 1993; Downs & Thorne 1998). Loss of meanders and reduced interaction with floodplains due to straightening and embankments reduce fish and wildlife habitat and destroy the complex food webs supported by floodplains (Iversen *et al.* 1993; Pearce 1993; Toth 1993; Shields *et al.* 1997). Channel deepening results in a reduction in the amount of light reaching the riverbed and so rooted plants cannot grow (Brookes & Hanbury 1990). Excavation or bulldozing of substrata from the riverbed reduces heterogeneity of the hydraulic biotopes available to aquatic organisms, with natural geomorphological patterns, such as pool-riffle-sequences being lost

(King & Schael 2001). Disruption of the substratum may also kill benthic organisms, which are an important source of food for fish (Brookes & Gregory 1988).

Rutherfurd *et al.* (2000) outline how maintenance practices can impact on diversity. Regular disturbances reduce habitat diversity and limit biota diversity within and along the channel. Snags provide shelter from fast flow, as well as shade, feeding and spawning sites, nursery areas for juvenile fish, territory markers and refugia from large predators and a variety of microhabitats within the complex surface structure of the snag. The microhabitats host a diverse range of invertebrates, algae and microbes, which play an important role in the maintenance of water quality of the system as well as being a major food source for organisms higher up the food chain. Snags also provide habitat for reptiles, birds and mammals that use the stumps as nesting, foraging and lookout sites. Their role in habitat provision is especially important in sandy rivers where they provide the only hard substratum available for colonisation. The removal of snags and other in-stream obstructions result in a loss of these habitats to aquatic organisms. Bank maintenance, where woody and herbaceous vegetation is removed or mowed result in poor habitat diversity and a uniform mix of species along the channel (Downs & Thorne 1998).

3.4.3 Canalisation

The term canalisation is not commonly used in the international literature. Canalisation can be defined as channelisation where the bed and bank are lined with concrete. It is most commonly used for flood control in urban areas. Concrete lined channels have smooth surfaces, solid walls and a uniform shape. The solid walls eliminate bank and bed erosion, while the smooth surfaces and uniform shape convey floodwaters efficiently.

Canalised rivers in South Africa include the Liesbeek River (Luger 1998), Kuils River (Ninham Shand & Chittenden Nicks 1999) Sand, Keyser, Diep and Black Rivers (Brown 1992) in the Western Cape; and the Apies and Jukskei Rivers in Gauteng (Garner 1997). Most rivers in and around Cape Town are canalised and have few remaining ecological attributes, now merely acting as conduits for storm water runoff and sewage effluent (Brown 1992).

Ecological impacts of canalisation

Concrete lining of channels severely impairs their ability to support life. The concrete lining eliminates all riparian, marginal and rooted aquatic habitats for plants. Riverbeds and banks lose all physical heterogeneity, and so they provide no substrata, food or refugia for aquatic animals. Canalised rivers are usually isolated from their catchments by their artificial channels, and so receive no surface or groundwater inflows except via drains. They also provide no input to groundwater and so the water table beneath them may drop (Luger 1998). Reduced biological activity limits the assimilative capacity for pollutants, and so high pollutant loads draining in from urban catchments are not materially processed or decomposed (Brown 1992; Luger 1998).

Floodplains outside the channels will tend to become terrestrial, as canals are designed to carry the full range of low and high flows (Toth 1993). Thus, floodplain refugia and nursery areas for fish are lost as well as the interchange of nutrients and sediments between floodplain and river (Schropp &

Bakker 1998). Floods greater than the designed capacity will still overtop the banks, but will probably flood riverside developed areas rather than floodplains.

3.4.4 Instream mining

River channels are mined of their substrata to obtain raw material for construction. Mining changes channel morphology in the same way as deepening, by changing the hydraulic geometry of the channel. For example, in-channel gravel mining for building material was responsible for changes in the long profiles of the Drôme River and its tributaries (Landon *et al.* 1998). The local risk of over-bank flow was reduced by gravel extraction, but flooding worsened downstream probably due to increased deposition of disrupted sediments reducing the channel capacity. In-stream gravel mining accelerated and worsened a trend of reduced sediment transport in the Drôme on a watershed scale (Landon *et al.* 1998). Incision in the mined reach exceeded 5 m with 38% of its total length being affected by incision. Gravel and sand mining, together with channelisation and upstream water abstraction, deeply incised the Grensmaas River in the Netherlands (Silva & Kerkhofs 1994). The river now only inundates its floodplain during very high floods, and carries very low flow during the dry season due to lack of groundwater inflow from floodplains that are no longer replenished. Mining also lowers the water levels by lowering the riverbed. Water levels in the Middle and Lower Rhine Rivers near Duisburg were lowered by about 2.5 m, partly due to mining (Admiraal *et al.* 1993).

In-stream mining has the same ecological impacts on the river as Deepening (Section 3.4.1).

3.5 Summary

Catchment hillslopes are formed by weathering and removal of rock and soil, by three main processes that operate at different spatial and temporal scales. The processes are mass wasting, solution and soil erosion, of which the latter is considered most relevant in terms of anthropogenic impacts. During soil erosion, water is the most common agent by which the soil is transported. The rate of infiltration of water into the ground, and the rate of interception of rainfall by vegetation, are important variables that influence the amount of water available on the ground surface for transporting soil. These and other factors contributing to the rate of soil loss can be summarised using the Universal Soil Loss Equation. This recognises the characteristics of climate and raindrop erosion, soil, topography and vegetation as factors affecting the rate of soil loss. Since there is local variability in these factors across a landscape, soil erosion and other hillslope processes vary across catchments. Land-use differences within a catchment also affect the erodability of these different areas.

In-channel disturbances are associated with a range of human activities, including damming, degradation or aggradation of sediments in the channel due to activities in the catchment, channelisation or canalisation.

Channelisation can involve straightening, deepening, widening or narrowing the channel, maintaining the banks in some desired (unnatural) form, or erecting embankments. Current speeds are increased when the channel is straightened or narrowed, leading to local and upstream scouring of the bed and deposition of eroded sediments downstream. Deepening and widening the channel result in an increased channel capacity, a decrease in current speeds and a subsequent increase in deposition.

Canalisation is an extreme form of channelisation, whereby the channel bed and banks are lined with concrete. Instream mining has the same impacts as deepening.

The ecological repercussions of channelisation are many and complex. The hydraulic regime and geometry of the channel are changed, with the channel heterogeneity caused by natural flows replaced by homogenous trapezoidal channel forms with near-uniform flow. This loss of heterogeneity inevitably translates into a loss of ecological biodiversity. Rivers become isolated from their banks, floodplains and groundwater, eradicating the interdependence of these in terms of the movement of sediments, water and nutrients. Maintenance of rivers, through mowing and dredging, prevents plants from establishing on the banks and within the channel, while removing natural instream features eradicates habitat for a range of aquatic and terrestrial species.

4. DEGRADATION: HYDROLOGICAL CHANGE

4.1 Introduction: catchment hydrology

Hydrology is the scientific study of water in the hydrological cycle. It is a discipline that studies the properties, occurrence, distribution and movement of surface and underground water. Hydrological processes can be divided into two categories: those with a lateral dimension, e.g. movement of water from land into channels; and those with a longitudinal dimension, e.g. stream-flow in channels (Fogg & Wells 1998). Water is moving all the time, changing from liquid to gas or solid states not only in the atmosphere but also in the soil and exposed water surfaces (e.g. streams, wetlands, oceans). The hydrological cycle encompasses the continuous transfer of water from the atmosphere to the ground through precipitation (rain, hail, snow), its runoff to rivers and storage in lakes and wetlands, and its return to the atmosphere through transpiration of plants or evaporation from interception sites, soil surfaces and open water bodies. Evaporation can be defined as the process whereby water is changed from a liquid to a gas, water vapour, and transferred to the atmosphere. The condensation of water vapour results in precipitation via cloud formation (Strahler 1969; Davies & Day 1998; Fogg & Wells 1998). Precipitation not intercepted by vegetation or flowing as surface runoff, infiltrates or percolates through the soil and becomes part of the soil water (groundwater) as it fills available spaces between soil and rock particles (Chapter 3). Soil water may be returned to the atmosphere from plants that absorb it through their roots and discharge it in the form of water vapour, through their leaf pores (stomatas), a process called transpiration. The term evapotranspiration is used for the combined water loss from direct evaporation and transpiration from vegetation (Strahler 1969; Fogg & Wells 1998).

The entire land area that delivers water, sediment and dissolved substances via surface or groundwater to a river and ultimately to the sea is called the watershed, drainage basin or catchment of that river. Any change in a catchment that effectively alters the natural drainage system, eliminating any of the storage areas or flow routes, will change the hydrological cycle within that catchment (Morisawa 1985; Fogg & Wells 1998). Most hydrological research is done at the catchment scale, providing data and understanding of water balances, flow extremes and effects of land-use and channel changes (Morisawa 1985).

According to Rowntree (2000) the catchment is a spatially defined system that converts precipitation into streamflow. There is a strong interaction between the hydrological regime that shapes the channel network in the long term, and the geomorphological structure of that channel network within the catchment that determines the streamflow response (Rowntree 2000). Different climates result in different patterns of catchment runoff. Factors such as catchment geology, topography, lithology, drainage basin size and vegetation, however, are important locally. All the above factors interact to determine the seasonal pattern of runoff in any one river. Globally, most regions show some degree of seasonality in rainfall and river flow regimes. The Department of Water Affairs (1986) stated that rainfall in South Africa tends to have a definite seasonality, which is reflected in the streamflow. Catchment characteristics link with climate to influence the degree of perenniality of a river and the flashiness of its flow regime (Petts & Foster 1985; Fogg & Wells 1998; Rowntree 2000). Catchments with wetlands and ample groundwater storage tend to have perennial rivers, and steep headwater streams tend to have flashier flow regimes than flatter lowland rivers.

Humans have intervened in every part of the hydrological cycle, through their various uses of water. Through changing water movement and quantity in the various components of the hydrological cycle, they have affected not only the hydrological cycle in its wider sense, but also the flow regimes of the catchment's network of surface waters and thus the ecological nature and functioning of those streams (Newson 1994; Watelet & Johnson 1999).

Physical disturbance of the catchment and its impacts on rivers have been discussed in Chapter 3. This chapter addresses interventions that directly manipulate the water in rivers, as well as a major indirect intervention. Direct manipulations are generally related to engineering schemes designed to alleviate water shortages or address existing or future threats of flooding, sedimentation or erosion. They may be categorised as follows:

- abstraction of surface water and groundwater;
- construction of dams and the consequent regulation of the river's flow regime;
- import or export of water via inter-basin transfer schemes (IBTs) and the use of channels as supply conduits;
- irrigation return flows from agricultural fields;
- discharge of effluents into watercourses.

The indirect intervention addressed in this Chapter is catchment hardening. This occurs mainly through the construction of buildings and roads in urban areas.

These interventions, together with other physical disturbances that impact the hydrological cycle (Chapter 3), have led to a global trend of increasing modification of the flow regimes of rivers. In individual rivers, specific parts of the flow regime may have been removed due to damming or abstraction, artificially generated through release of dammed water or water imported from another catchment (Smakhtin & Watkins 1997) or exaggerated due to catchment hardening. In this Chapter, dams and reservoirs are addressed in Section 4.2, inter-basin transfers of water in Section 4.3, direct abstraction of run-of-river flow and groundwater in Section 4.4, agricultural return flows in Section 4.5 and catchment hardening in Section 4.6.

4.2 River regulation: dams and reservoirs

There are more than 40 000 large dams worldwide (Davies & Day 1998) and an uncounted number of smaller ones. Artificial releases into the downstream river can be made from large dams, through specially constructed outlets, but small dams have little or no control over their level of storage and releases (Smakhtin 2001). Dams now regulate flow in the majority of the world's rivers, in order to meet the high and growing demand for reliable sources of water. Water stored in and released from dams is used not only for water supply, but also for hydropower generation, irrigation, flood control, recreation, and to aid navigation along rivers (Dynesius & Nilsson 1994; Shuman 1995; Miller 1996a; Born *et al.* 1998). Rivers are recognised as among the world's most severely degraded ecosystems, and globally, flow regulation through dams is probably the single most significant impact of humans on rivers (Dynesius & Nilsson 1994; Shuman 1995; Ward & Stanford 1995; Rapp 1997).

Dams usually store small to medium size floods as well as parts of larger floods and some of the low flow. The stored water may be released downstream at some later stage or diverted away from the river. In both cases, downstream river ecosystems are impacted, either by a permanent reduction or enhancement in the number and magnitude of different flows that created them, or by the pattern of high and low flows being changed. Changing the flow pattern results in high or low flows arriving at a time of year when they would naturally have been rare. Such flows may also be different than historical flows in terms of temperature, water quality or the amount and kind of sediments carried. These changes will affect the downstream river ecosystem, in the ways outlined below.

Ranging from small temporary structures to major multi-purpose dams, in-channel barriers affect the physical and hydrological environment of the channel, the riparian zone, associated wetlands, floodplains and the estuary. The various impacts of a dam depend on the purpose for which it was built, its structure, its size in relation to stream flow, its location and its operational procedures for storing and releasing water. Effects may be manifested many kilometers or tens or hundreds of kilometers downstream from the dam (Newson 1994; Fogg & Wells 1998).

Petts (1980, 1984, 1988) provided a useful theoretical framework for the impacts of dams. He suggested that there are three orders of impact (change) within a river system after dam construction:

- first-order impacts involve changes in the flow regime and sediment load (process alteration);
- second-order impacts involve a resulting change in channel morphology and abiotic (non-living: physical and chemical) habitats;
- third-order impacts reflect the first and second-order changes, through changes in the biotic (living) components of the system.

First-order impacts

There are several common effects of dams on the flow and sediment regimes of downstream rivers. First, flood peaks, and hence the frequency, magnitude and duration of overbank flooding, may be reduced (Petts 1980 1984; Williams & Wolman 1984; Davies *et al.* 1993; Nilsson & Berggren 2000). Ligon *et al.* (1995) note, for example, that since the construction of two flood-control dams in the 1960s, the McKenzie River in Oregon is gradually changing because of the reduction of peak flows. In the past, peak flows reached 1100-1800 $m^3 s^{-1}$, but now reach only to 500-800 $m^3 s^{-1}$ (approximately bankfull discharge). Regulation since 1966 on the Gunnison River in Colorado has resulted in the reduction of flood magnitudes of given recurrence intervals (Elliott & Parker 1997). The ten-year flood decreased from 422 to 198 $m^3 s^{-1}$, the five-year flood from 360 to 155 $m^3 s^{-1}$ and the mean annual flood from 263 to 114 $m^3 s^{-1}$. The pre-dam bankfull discharge (recurrence interval of 1.25 years) of 165 $m^3 s^{-1}$ decreased to 62.3 $m^3 s^{-1}$after dam construction. Low and moderate stream flows were augmented however. The duration of flows between 32.3 and 85.0 $m^3 s^{-1}$ increased from 12% to 38% of the time. Vaselaar (1997) and Schmidt *et al.* (1998) demonstrated that the Glen Canyon Dam and its operations reduced the frequency, magnitude and duration of floods of the Colorado River as it flows through the Grand Canyon National Park to Lake Mead and the Hoover Dam. Before dam construction the magnitude of the two-year recurrence flood for the period 1921-1962 was 1250 $m^3 s^{-1}$ and flows that exceeded this were sustained for 30 days or more. After construction of the dam in 1963, dam releases were characterized by large, hourly fluctuations resulting from load-following hydroelectrical power production.

Second, seasonally varying flows may be reduced, increased or reversed, masking regional flow differences. Many natural rivers have periodic, often seasonal flows (Stanford *et al.* 1996; Davies & Day 1998; Sparks *et al.* 1998; Nilsson & Berggren 2000). In South Africa for example, summer high and winter low flows are found in the summer rainfall regions (located in the interior and eastern plateau slopes of the country) while winter high and summer low flows are found in the south-western winter rainfall region. Non-seasonal patterns of flow are found in the areas between (South African Department of Water Affairs 1986; Davies *et al.* 1993). Downstream variation in flow can be reduced where dams are operated for flood control, or increased where they are used to create hydropower (Johnson *et al.* 1995). In extreme cases the seasonal flow pattern may be reversed, such that the dam harnesses high flows in the wet season causing unnaturally low flows downstream, followed by unnaturally high flows in the dry season as the stored water is released downstream for irrigation or other types of demand. For example, Davies & Day (1998) noted that releases from the hydroelectric power-producing Gariep Dam on the Orange River reach a maximum in the winter even though the highest discharges in the river naturally occur in the summer. Elliott & Parker (1997) working on the Gunnison River demonstrated that upstream dams altered the seasonality of the river. The monthly mean discharge from April to July decreased by 63% after dam construction, whereas for the remainder of the year, August through March, it increased by 170% (Elliott & Parker 1997). By 1967, six dams had been built on the middle reaches of the Missouri River, Missouri, operating as a system to support navigation on the lower river. In order to maintain the channel depth for April-November navigation, the March and June flood pulses were depressed and the late summer/autumn low flows were augmented (Galat *et al.* 1998). Sparks *et al.* (1998), demonstrated how dams can mute downstream flow variability. They studied 28 low navigation dams built on the Mississippi River USA during the 1930s. These dams, only 2.7 m high, do not harness large or medium floods, but by not allowing the water level to drop as low as it used to, they have reduced the range of natural water level variation in the river. The flow regulation resulted in riverine organisms having to cope with flows that never reach historical low levels, and which almost never reach flood level (Davies & Day 1998).

Third, dams retain sediments that would normally move downstream. The entire bedload moving along the river, as well as most of the suspended load, may accumulate within the reservoir (Petts 1980; Heede 1986; Sear 1992; Ho 1996; Davies & Day 1998). Sediment deposition in dams can lead to increased floods upstream, loss of water storage and thus loss of yield, blockage of intakes, abrasion of turbines and downstream problems involving both aggradation and degradation. Sediments settling in the dam may eventually cause upstream channel changes because the level of the riverbed at the upstream end of the reservoir may become raised. This raise may lead to aggradation and possible localized flooding events upstream (Maddock 1999; Heede 1986; Nilsson & Berggren 2000). Vaselaar (1997) and Schmidt *et al.* (1998) for example, showed that the average suspended sediment load of the Colorado River before the construction of the Glen Canyon Dam was approximately 6.0×10^{10} kg yr^{-1} at Lees Ferry, and 0.000013×10^{10} kg yr^{-1} after completion of the dam. Sutadipradja & Hardjowitjitro (1984) reported that reservoirs in Indonesia are silting up and their lifetimes are being shortened. The Seloredjo Dam in East Java and the Cacaban Dam in Central Java have a siltation rate of 209 and 218 mm yr^{-1} respectively and they annually lose 1.2 % and 2.4 % of their capacity. The Aswan Dam on the Nile River in Egypt traps large volumes of sediments, resulting in major downstream geomorphological and ecological impacts. As a result the fertile Egyptian Nile delta agricultural lands no longer receive nutrient-rich sediments and are being eroded and transported into the Mediterranean Sea (Miller 1996a).

Surian (1999) showed that hydroelectric dams built between 1930 and the early 1960s on the Piave River in the Eastern Alps, Italy, have altered the sediment supply of the river. After completion of the dam trapped sediment was in the excess of 1×10^6 m^3 yr^{-1}. Today it is estimated at 145 000 m^3 yr^{-1}.

Water released from such dams is likely to contain very low sediment concentrations and may be "sediment hungry", (Petts 1980; Williams & Wolman 1984; Heede 1986; Ligon *et al.* 1995; Ho 1996; Vaselaar 1997; Erskine *et al.* 1999; Nilsson & Berggren 2000) and so scour or erode sediments from the channel's bed and banks downstream from the dam. The degree of scouring will depend on the nature of the bank and bed materials, with bedrock rivers changing least and alluvial rivers more as their beds and banks are scoured of finer sediments. Scouring of fines armours the riverbed, with the composition of the remaining sediments coarsening, and the channel gradually adjusting in other ways (Davies & Day 1998). Williams & Wolman (1984) pointed out that degradation is most profound near the dam and diminishes downstream. It proceeds at high rates during the first 10-20 years after dam closure and decreases thereafter.

Second-order impacts

According to Petts (1980), the simplest reaction to a changed flow regime is that the new flow regime is accommodated within the existing channel. This occurs where the channel boundaries are too stable to allow any change. The reduced flow is simply accommodated within the existing channel form and has sufficient power to transport available sediment through the channel. Ligon *et al.* (1995) were able to show that stability may be further enhanced by established stands of vegetation.

Numerous other studies (Gill (1973), Buma & Day (1977), Petts (1980, 1988), Dynesius & Nilsson (1994), Ligon *et al.* (1995), Williams (1996), Elliot & Parker (1997), Fergus (1997), and Friedman *et al.* (1998)), have however revealed extensive alteration and degradation of river ecosystems by dams. Depending on the type of river and type of dam, a range of geomorphic changes may occur downstream of dams in response to changes in flow and sediment regimes. The exact nature of these changes and their magnitudes and timing often remain unpredictable, however, channels commonly becoming incised, aggraded, or simplified (Petts 1980; Moffat 1993; Ligon *et al.* 1995). Ligon *et al.* (1995) reported that a hydroelectric dam constructed in the 1950s on the Oconee River in the USA resulted in channel incision. The river lowered its bed and deepened its channel by approximately 1 m, with coarsening of sediments particularly apparent on the surface layer of in-channel sediment deposits. In response to the new lower level of the Oconee River, tributaries incised their channel beds, which in turn resulted in terraces on tributaries near their confluence with the Oconee River. Ligon *et al.* (1995) and Brizga *et al.* (1999b) suggested that armouring and tributary incision are common responses to river incision. Additionally incision can lead to a lowering of the ground water table underlying the riparian zone and reduction in the extent of the active floodplain, leading to changes in the composition of riparian species on the floodplain (Nilsson & Berggren 2000). Within nine years of closure of the Hoover Dam on the Colorado River, for instance, the river downstream of the dam cut down 4 m and lost more than 110 million m^3 of sediment in the first 145 km after the dam (Davies & Day 1998). Time spans of 180 -200 years may be needed before a new equilibrium on the Colorado River is reached (Heede 1986).

Reduction in peak flows and sediment loads may lead to channel simplification in rivers. Examples of channel simplification include reduced river meandering, reduced deposition on bars and the conversion of braided channels into single-threaded channels (Ligon *et al.* 1995; Friedman *et al.* 1998; Stewardson *et al.* 1999; Nilsson & Berggren 2000). Where there are meandering channels, dams lead to reduction in the rates of channel migration and increases in the establishment of riparian vegetation. Rates of bank erosion and bar deposition are also typically reduced (Johnson 1992; Friedman *et al.* 1998; Nilsson & Berggren 2000). Dam-regulated flows on the Colorado River have led to a reduction in the size of sandbars, an altered riparian ecosystem, a build up of debris fans and sedimentation of backwater areas used by native fish (Vaselaar 1997; Schmidt *et al.* 1998). Ligon *et al.* (1995) noted that reductions in peak flows are changing the Mckenzie River, through stabilization of the channel, as the flows no longer erode or cut against its banks. Mid-channel bars and islands are not being created and maintained, and secondary channels are filling, changing the channel pattern to a deep and narrow single-thread (Ligon *et al.* 1995). Changes in channel pattern take decades, and so the narrow single-thread channel may be an intermediate stage in adjustment (Petts 1980; Ligon *et al.* 1995; Stewardson *et al.* 1999).

The composition of the streambed substrata may become coarser or finer (Petts 1980; Heede 1986; Stewardson *et al.* 1999), and there could be increased or decreased lateral migration of channels (Petts 1980; Ligon *et al.* 1995). The lack of large flooding events may lead to the accumulation of coarser sediments and aggradation on the channel bed and banks, depending on the availability of sediment (Petts 1980; Sear 1992). Sear (1992) provided evidence that the Kielder Reservoir completed in 1981 on the River North Tyne, Northumberland, U.K., trapped the sediment supplied from the largely afforested catchment area but also resulted in siltation and coarsening of downstream riffle areas. There was an increase in fine sediment (< 1 mm) at 64% of riffles up to 15 km downstream from the dam while surface coarsening of clasts (> 64 mm) was apparent up to 6 km from the dam.

The channel can become wider or narrower (Erskine *et al.* 1999; Brizga *et al.* 1999b; Surian 1999), and vegetation in the riparian zone can be reduced or can encroach into active channels (Gill 1973; Nilsson 1996; Kondolf 1998). Brizga *et al.* (1999b) demonstrated that the Yarra Dam, completed in 1957 on the Yarra River in Victoria, Australia, resulted in reduced flows which in turn caused vegetation encroachment. Colonizing species were weeds such as willow trees rather than indigenous riparian species. The encroachment of vegetation can promote the deposition of fine sediments and increase bed and bank resistance to erosion thus resulting in channel narrowing. In the Yarra River significant channel narrowing became increasingly apparent because of the delivery of sediments from tributaries. This kind of channel change can take from decades to centuries, depending on the availability of sediments. Narrower channels have a reduced capacity to carry floods and so are usually accompanied by a rise in flood levels (Brizga *et al.* 1999b).

Research on the geomorphological impacts of dams on downstream rivers is scarce in South Africa, although it began to increase in the early 1990s. Dollar (1990) demonstrated that the Sandile Dam on the Keiskamma River resulted in the attenuation of flood peaks and the deposition of bars at downstream tributary confluences. There were also sedimentation problems downstream of the dam. Research by McGregor (1999) on the Keiskamma River also revealed bed degradation, the siltation of a side channel 100 m below the Amatola junction, and development of a tributary bar at the Ncqwazi and Keiskamma junction.

Third-order impacts

The literature on downstream ecological impacts of dams is extensive and cannot be fully reviewed here. Readers are advised of McCully's (1996) comprehensive review of the subject as well as the Final Report of the World Commission on Dams (2000). Here, some major impacts of dams are briefly summarised.

Abiotic changes in the downstream river following dam construction induce a range of responses in the riverine biota. In general, changes in physical, chemical and thermal conditions may eradicate or fragment instream habitats, eradicate vital linkages between the river and floodplain or riparian zones, and block migration pathways. As a result, species may be eliminated or reduced and others gained (Petts 1980; Ligon *et al.* 1995; Stanford *et al.* 1996; Nilsson *et al.* 1997; Born *et al.* 1998). Exotic invasive species often become established in such disturbed areas, and may out-compete remaining native species leading to their further decline.

For example, fish communities in a dammed river can undergo drastic changes due to disturbances to fish passage. Adult fish may not be able to migrate upstream past high dams to spawn, and juvenile fish may take far longer than natural to migrate downstream because of having to traverse deep, slow-moving reservoir water. This exposes them to predators for a longer time (Fogg & Wells 1998; Rutherfurd *et al.* 2000). Fish numbers may be further reduced by the elimination of high flows that maintained gravel spawning-areas free of silt (Ligon *et al.* 1995; Fogg & Wells 1998; Vehanen & Riihimäki 1999).

Dams can create severe water-quality problems, such as nutrient-poor water, or nutrient enrichment that can result in excessive growth of plants, including algae. Eutrophication, defined as the excessive supply of nutrients to water from point or non-point sources, can result when reduced flows occur in rivers receiving high nutrient loads. In the worst cases, eutrophication can result in toxic algal blooms, which can in turn lead to the loss of a wide range of aquatic flora and fauna (Wiechers *et al.* 1996; Clarke & Wedderburn 1999; Fogg & Wells 1998; Vehanen & Riihimäki 1999). On the other hand, nutrient-poor water can result from sediments being trapped by dams. Organic material, which provides vital nutrients, may be trapped with the sediments, to the detriment of downstream aquatic communities (Fogg & Wells 1998).

Changes in water temperature imposed by dams may affect downstream aquatic communities in a variety of ways. Constant releases, for instance, may create more constant temperatures, affecting those communities dependent on temperature variations for reproductive cues. Low flows warm quickly and hold less dissolved oxygen than cooler water, contributing to stress and deaths of many organisms (Shuman 1995; Fogg & Wells 1998; Schmidt *et al.* 1998). Large storage reservoirs with little water movement may have water at deeper levels that is cooler than the inflowing river. Release of this abnormally cold water can kill existing fish larvae, delay further spawning and threaten the survival of late-hatched juveniles (Cambrey *et al.* 1997; King *et al.* 1998). Water-quality changes are further elaborated in Chapter 5.

Dams prevent or at least regulate downstream flooding and the interchange between river and floodplain, altering instream and floodplain habitats (Ligon *et al.* 1995; Nilsson *et al.* 1997; Galat *et al.* 1998). Normally the floodplain is inundated for far less time than before dam construction. A reduction in the aerial extent and duration of floodplain inundation often results in decreased species diversity (Ligon *et al.* 1995). Dams also harness the small and medium floods that appear to maintain the different zones of vegetation up banks (Boucher 2002), reducing the extent and diversity of these zones. Reduced flooding can lead to the gradual migration of vegetation zones downward toward the active channel, narrowing it, so that there is less capacity to carry the large floods that the dam cannot store (Petts 1984; Stanford *et al.* 1996; Nilsson *et al.* 1997; Brizga *et al.* 1999b). The general trend is that dams create a discontinum of environmental conditions, and habitats for riverine organisms become homogenous, limited by the conditions dictated by the operation of upstream dams (Stanford *et al.* 1996).

Native biodiversity almost always decreases in dammed rivers, while exotic species proliferate. Introduced species may compete directly with native species, they may predate on them or they may alter their habitats (Stanford *et al.* 1996; Stange *et al.* 1999). In some cases, however, productivity is increased by flow regulation, although the composition of species will change, usually with the loss of rare and sensitive species (Chutter & Heath 1993).

4.3 River regulation: inter-basin transfers (IBTs) and diversions

Transfer of water from areas of surplus to areas of deficiency has become a common solution for securing water supplies for human settlements (Petitjean & Davies 1988; Snaddon & Davies 1998). Davies *et al.* (1992) defined an inter-basin transfer of water (IBT) as the transfer of water from one geographically distinct river catchment or basin or reach to another. IBTs developed in response to population increases, industrialization, urbanization and agricultural water demands, and occur mainly in areas where water demands exceed local supply. In many rapidly developing countries, such as South Africa, as well as in developed dryland regions, water transfers have become the lifeblood of development and future needs (Davies *et al.* 1993; Snaddon *et al.* 1998).

In the past, there has been little research on the impacts of IBTs on the donor or receiving river systems. Although some IBTs were technically assessed, worldwide, their ecological and geomorphological impacts were virtually ignored or underplayed. Comparative research of the pre-transfer and post-transfer conditions of the rivers has rarely been undertaken, implying that IBTs are being planned despite lack of knowledge of their environmental implications. In the last two decades, however, there has been growing concern over the detrimental impacts of IBTs to both donor and recipient rivers (Davies *et al.* 1993; Petitjean & Davies 1988).

The nature of the components of an IBT will influence the extent of its effects on the rivers. Such components include the arrangement in the donor system (direct extraction from the river, or via a dam or weir); the arrangement in the receiving system (directly into the river or into a reservoir); the transfer route (buried pipes or open canals), and the operational criteria (e.g. the type and timing of release, and the volume and rate of water transfer). The water is normally released from a pipeline, canal or tunnel and is often under pressure due to pumping or gravity. The timing of release can be seasonal or aseasonal, constant or pulsed, or some combination of these (Davies *et al.* 1992; Snaddon *et al.* 1998). Generally the effects of dam-facilitated diversions on donor streams are similar to those caused by dams in general (last section), but there are additional, often severe, effects on recipient rivers. For instance, the imported water may have a different temperature or water quality than those of the receiving river, or it may introduce biota that does not naturally occur there. This can cause problems not historically associated with the receiving catchment (Snaddon & Davies 1997; Snaddon *et al.* 2000).

4.3.1 Geomorphological effects of IBTs on riverine ecosystems

IBTs lead to changes in the quantity and flow regime of water in both the donor and recipient systems. Flow regimes may change from seasonal to perennial or vice versa (Snaddon *et al.* 1998; Snaddon *et* *al.* 2000). Increased flows in the receiving system can result in erosion and incision of the channel bed, enhanced instability of banks, armouring of the streambed, channel enlargement, the loss of pool-riffle sequences or the lowering of high riffles (Ryan 1997; Dominick & O'Neil 1998; Snaddon *et al.* 1998; Broderick & Outhet 1999; Erskine *et al.* 1999; Snaddon *et al.* 2000). Enhanced deposition of these eroded sediments will occur downstream. The donor river will experience reductions in flow, which can manifest as decreased base flow or reduced flood peaks, with a resulting reduction in flushing flows, sedimentation and channel narrowing (Ryan 1997; Snaddon *et al.* 1998). See Section 4.2 and Section 4.4 for further information on the effects of IBTs on donor systems.

4.3.2 Ecological effects of IBTs on riverine ecosystems

The ecological effects of IBTs are site specific, varied and very complex. The major effects are in the receiving system, manifested as changes in water quality and quantity, and in responding changes in the species composition of aquatic plants and animals.

Donor systems may have different water quality than the receiving streams, resulting in the latter developing chemical regimes significantly different from their natural ones (Snaddon *et al.* 1998). Water-quality variables that may change include conductivity (salinity), turbidity (Miller 1996a; Snaddon *et al.* 1998), pollutants, nutrients and temperature (Snaddon *et al.* 1998; Snaddon *et al.* 2000).

Increased salination can occur in either the donor or recipient system. For instance, after becoming a donor catchment, salinity in the Aral Sea tripled, resulting in the elimination of all native fish and half of the birds in the area (Miller 1996a). Receiving systems can similarly be affected. Water transfers ($157 \times 10^6\ m^3\ yr^{-1}$) from the lower reaches of the Murray River in South Australia have increased the salinity and turbidity of all the river's recipient systems. Nutrient levels in the recipient reservoirs were increased and resulted in the increase of blooms of toxic cyanobacteria. Other ecological effects of increased turbidity and salinity are not known for these freshwater systems (Hart 1992; Snaddon *et al.* 2000). However, some receiving systems show the opposite result through a dilution process. The upper reaches of the Great Fish River (South Africa) became less saline (particularly with regard to sodium, chloride, magnesium and sulphate) when water was introduced from the Orange River. The number of taxa of invertebrates changed little (from 41 before the transfer to 47 after), but the species composition changed considerably. There were significant shifts in the dominant Chironomidae (ghost midges), hydrosychid Trichoptera (caddisflies) and Simuliidae (blackfly) species. All the shifts in the riverine invertebrate species could be directly linked to changes in the flow regime (the river is now perennial rather than seasonal), caused by the transfer (Petitjean & Davies 1988; Snaddon *et al.* 2000).

IBTs can dilute point-source pollutants in receiving systems, thereby reducing their ecological impacts. More likely, however, donor systems may introduce water of inappropriate quality to the recipient system, such as warm water from storage dams or lower reaches into cool headwater streams (Snaddon *et al.* 1998; Snaddon *et al.* 2000). Temperature changes can affect riverine systems in the following ways: some species may become excluded from the effected area, water-quality parameters may alter (e.g. increased temperature can result in a reduction of dissolved oxygen), timing and development of life cycles may be affected (e.g. changing the timing of insects emerging from a river) and algae and other plant growth may change (Rutherfurd *et al.* 2000). All these changes impact the riverine biota, causing stress, behavioural changes or even death. Eutrophication in donor systems can result in the development of algal blooms that are transferred to receiving systems. This occurred in the Western Cape, where geosmin was introduced from Theewaterskloof Dam on the Breede River system to the Berg River. The presence of geosmin in the Berg River has tainted the flesh of rainbow trout reared in a commercial farm on the river. Geosmin imparts a plastic-like taste and odour to drinking water (Davies & Day 1998; Snaddon & Davies 1998).

IBTs may have a significant impact on aquatic and riparian vegetation, because of the interrelationships between riparian vegetation, hydrological regime and channel morphology. Both vegetation increases and losses have been recorded in receiving systems (Stormberg 1993; Collett 1996; Johnson 1998), and sometimes increases and losses can occur simultaneously in different reaches of the same receiving system. Receiving systems may also be subjected to increased bank erosion and the loss of indigenous vegetation due to inundation or, alternatively, a perennial availability of water may lead to the invasion of riparian zones by alien vegetation and an increase in the area of riparian vegetation (Stormberg 1993; Collett 1996; Johnson 1998). Riparian vegetation in donor systems is likely to be reduced due to the lower base flows (Collett 1996; Johnson 1998; Snaddon *et al.* 2000).

Loss or modification of aquatic habitat may occur in both donor and recipient systems, due to the changes in flow and channel morphology (Snaddon *et al.* 1998). This may have major effects on the biotic assemblages. For instance, species composition of the aquatic plant and animal communities in the receiving system may change if flows are more constant, with new dominant species. This can result in unwelcome changes in the river, with favoured species declining in abundance, and new species, possibly pests, appearing (de Moor 1994). Additionally, native or exotic species introduced from the donor system may cause disruptions in the receiving system. Species alien to a receiving system will compete with its native species for nutrients, moisture, sunlight and space, and can impose widespread, intense and continuous stress on them. They may also be predators of the native species, hybridise with them, or introduce parasites and diseases (Fogg & Wells 1998; Snaddon *et al.* 1998; Snaddon *et al.* 2000).

In the donor system, suitable nesting and feeding areas for aquatic vertebrates such as fish may be reduced due to a drop in water levels, and migration routes may become blocked by IBT canals and dams. Invertebrate communities downstream of the IBTs may be reduced in abundance and diversity, with an increase in planktonic groups due to the absence of flushing flows (Fogg & Wells 1998; Snaddon *et al.* 1998; Snaddon *et al.* 2000).

For a more comprehensive discussion of the impacts of IBTs, readers are referred to Petitjean & Davies 1988; Davies *et al.* 1992; Davies *et al.* 1993; Snaddon & Davies 1997, 1998; Davies & Day 1998.

4.4 River regulation: direct abstraction of surface and ground water

Surface waters occur in rivers, and standing waters such as lakes and other wetlands. Aquifers are saturated subsurface zones of sediments through which groundwater flows. The upper surface of the saturated zone, known as the water table, acts as a dividing line between the lower saturated and upper unsaturated soils. Aquifers act as huge, slow-moving underground lakes, and are replenished naturally by precipitation. There is a greater volume of groundwater than surface water, but it is unequally distributed with only a small amount economically exploitable (Miller 1996a).

Groundwater seepage from aquifers maintains the perenniality of rivers during the dry season and, in turn, rivers replenish nearby aquifers during higher flows. This interaction is vital for sustaining the river ecosystem, both in terms of maintaining surface-water habitat for aquatic species and sub-surface water for riparian vegetation. Human demands for water are now threatening this balance: aquifers have always been seen as desired sources of water, with high-quality and abundant waters, and abstraction rates from them are rapidly increasing worldwide (Miller 1996a; Stange *et al.* 1999).

4.4.1 Geomorphological and ecological effects of direct abstractions on river ecosystems

Water withdrawals from aquifers lower the water table, which can lead to the river losing water to the aquifer (stream leakage) and a consequent reduction in stream flow. If the aquifer is adjacent to the coast, sea levels may stand higher than the declining water table, resulting in saline intrusion and a decrease in water quality (Keyser *et al.* 1999).

Large-scale withdrawals of groundwater may significantly affect the quantity and quality of hydraulically linked streams, through intercepting the water that would otherwise drain to the stream. Excessive pumping of groundwater, especially in the dry season, may result in groundwater-dependent river systems drying up (Kondolf *et al.* 1987; Kondolf 1996; Chen & Soulsby 1997; Parsons 2000). Such pumping can reduce river flow to the extent that the most basic ecological processes in channels are not sustained. For example, massive die-off of willows trees and other riparian vegetation, destabilization of riverbanks and significant erosion during moderate floods can result from nearby groundwater abstractions (Kondolf 1996). Bank erosion can result in loss of aquatic habitats by changing a single-thread meandering channel with pools and vegetated banks into a wide, shallow, braided channel with eroding banks (Kondolf & Curry 1986; Kondolf 1996).

Because of the tenuous visible links between groundwater and river, the ecological effects of groundwater abstraction on rivers are not well documented. They are known to exist, however, with Kleynhans (1992) recording the impacts on surface waters of groundwater abstraction in the Greefswald Aquatic System.

Direct run-of-river abstractions reduce the quantity of the remaining stream flow and can result in similar channel changes to those discussed in Section 4.2, albeit usually less severe because floods cannot be abstracted. Such abstractions can reduce aquatic habitat, however, and so affect survival of a range of aquatic species (Le Maitre *et al.* 1999; Fogg & Wells 1998). King & Tharme (1994) recorded flow in the Olifants River (Western Cape) drying up as they watched, due to upstream pumping by farmers, only to begin again later in the day. Such drying-out events, which probably regularly occur in South African rivers, render large parts of the riverbed unsuitable as aquatic habitat, because they are not consistently wet. They also eradicate areas of passage for fish to escape unsuitable conditions or predators.

4.5 River regulation: agricultural return flows

Intensification of agricultural practices has led to increasing use of fertilizers and pesticides, with resulting significant inputs of these pollutants into natural drainage networks. Irrigation return flows enter the rivers over wide areas of a drainage basin as non-point source pollution (Elliott *et al.* 1998), and also percolate deeply into the soil to become part of the body of groundwater. Generally, as the irrigated areas expand, irrigation return flows increase in volume and may contribute to a large proportion of a river’s water balance (Smakhtin & Watkins 1997). Smakhtin (2001) reported that up to 40% of water initially abstracted for irrigation returns to the river. If irrigation water is imported from outside the catchment, return flows contribute a substantial additional sub-surface drainage directly to the river channels or through "return" canals affecting base flow levels.

Irrigation return flows can also significantly downgrade the quality of stream flow (Smakhtin & Watkins 1997), and are a major cause of eutrophication of rivers. Dukhovny & Stulina (2001) reported that return flows in the Aral Sea Basin resulted in severe water-quality problems. Intensive development of irrigation from 1940-1990, and the construction of large-scale collector-drainage networks, resulted in increased return flows that have contributed 100-115 million tonnes of salt and other harmful components to the basin's two biggest supply rivers, the Amudarya and Syrdarya. The Aral Sea shore has a unique combination of delta, seashore and desert ecosystems, and supports a diverse biota that is now threatened. Outflows of the polluted waters into natural sinks and pasture areas have also resulted in landscape degradation and the propagation of insects and weeds (Dukhovny & Stulina 2001).

The magnitude of return flows will depend to a large degree on the mode of application of irrigation water (drip, centre pivot, flood or spray). As available resources become scarce, irrigation return flows may decrease significantly because of the use of more efficient application methods such as micro-irrigation (Mckenzie & Roth 1994).

4.6 River regulation: urbanisation, catchment hardening and effluent flows

Urbanization results from people moving from rural areas to urban centres in pursuit of better living conditions. Globally, urbanisation is increasing, with rapid growth of urban areas. Such areas consist mainly of impervious surfaces, such as roads and buildings, and a process of catchment hardening accompanies their growth. In this process, the surface of the land is increasingly sealed by the urban developments, and infiltration of rainwater into the soil reduced (Morisawa 1985; Fogg & Wells 1998). Catchment hardening changes the drainage pattern to urban streams in two main ways. First, water that cannot infiltrate has to be removed to avoid flooding. This is done via storm-water drains that conduct the water through buried pipes or surface channels to the nearest river (Morisawa 1985; Newson 1994; Whitford *et al.* 2001). Water in storm-water conduits drains to the river more quickly than that seeping through the soil, and so more arrives at the river during storms. This increased direct runoff into the river increases flood peaks. Second, the reduced infiltration into groundwater results in reduced low flows during the dry season, because there is less groundwater to sustain them. The combined result is a flow regime that is flashier and less reliable, with greater flood peaks of shorter duration and a general tendency toward less perennial flow in the dry season (Hasfurther 1985; Helfield & Diamond 1997; Elliot *et al.* 1998). Flashier flow regimes, with increased flood peaks and higher flow velocities, can result in bed and bank erosion. The subsequent deposition of this sediment downstream can cause further engineering and ecological problems (Morisawa 1985; Fogg & Wells 1998; Pizzuto *et al.* 2000; Whitford *et al.* 2001).

Urbanisation also decreases the quality of river water in two main ways (Wimberley & Coleman 1993; Newson 1994; Wilson *et al.* 1996; Helfield & Diamond 1997; Elliott *et al.* 1998, Winter & Duthie 1998; Koning & Roos 1999; Prat & Munné 2000). First, storm-water drains collecting water from urban areas may carry high loads of pollutants such as oil and litter, and industrial discharges and spills. Second, storm-water drains may be a part of sewage networks and by this or other means may transport sewage into rivers. Additionally, sewage-treatment facilities usually release treated effluent directly into streams. Another growing source of pollution is informal or squatter settlements, which generally have poor services with regard to sanitation and refuse removal, and so may contribute disproportionately in terms of contamination of both surface and ground waters. According to Fogg & Wells (1998) urban streams tend to have poor diversity of aquatic habitats and biotas. Often, the channel is artificially confined to a narrow strip of land, canalised or channelised and cut off from its floodplains and catchment. The ecological effects of this have been outlined in earlier parts of this Chapter.

4.7 Summary

Humans have intervened in all land-based parts of the hydrological cycle. Whilst dams represent the greatest intervention to rivers, IBTs, direct abstraction, agricultural activities and urbanisation each have their own unique impacts. In general, these interventions have changed the pattern of high and low flows and sediment regimes of rivers, which have in turn led to changes in channel morphology, water quality and the biota.

Flow regulation through the construction of dams is the single most significant impact of humans on rivers. The various effects of a dam depend on the purpose of the dam, its size, its location and its operational procedures for storing and releasing water. Petts (1980, 1984, 1988) established a useful framework for considering the impacts of dams. He makes the point that there are three orders of impact within a river system after dam construction. The first set of impacts involve changes in the flow regime and sediment; responding to this, the second set involve changes in channel morphology and abiotic habitat; and thirdly, in a further set of responses, the riverine biota changes.

The first order or flow changes may lead to reductions in the frequency, magnitude, duration of overbank flooding and seasonal flow variability. Reservoirs may trap sediment loads behind their walls and result in sediment-hungry water downstream of the dam. Second-order impacts may include channel incision, channel simplification, coarsening of the substratum, channel narrowing, vegetation reduction, vegetation encroachment and habitat change. Third-order impacts are typified by a reduction in species diversity.

IBTs involve the transfer of water from areas of surplus to areas of deficiency. Worldwide their ecological and geomorphological impacts were almost ignored or underplayed until recently. Dam-facilitated IBTs on donor streams generally have the same impacts as those caused by dams. However, diversions do have additional, often serious effects on recipient rivers. As well as changing the flow pattern, flow volume and sediment dynamics of the receiving stream, IBTs have a host of site-specific, varied and complex ecological consequences.

Abstraction rates from groundwater are rapidly increasing worldwide. Aquifers are saturated subsurface zones of sediment through which groundwater flows. Groundwater seepage from aquifers maintains perennial rivers during the dry season and in turn replenishes nearby aquifers during higher flows. Generally, withdrawals from aquifers result in a lowering of the water table, a consequent reduction in stream flow, possibly with an intrusion of saline water and so a decrease in water quality. The ecological effects of groundwater abstraction on riverine ecosystems are generally not well documented. Direct run-of-river abstractions reduce the quantity of the remaining stream flow and can result in similar channel changes discussed under dams, although changes would be less severe. Agricultural return flows almost always result in water-quality problems and are major causes of eutrophication of streams. As irrigated areas expand, irrigation return flows increase and may play an important role in a river's water balance. Little is known concerning both the geomorphological and ecological effects of return flows.

Globally, urbanisation is increasing, with rapid growth of urban areas. Urbanisation results in catchment hardening, a process whereby the surface of the land is sealed by urban developments and infiltration of rainwater into the soil is reduced. Urbanisation results in a flow regime that is flashier and less reliable, with greater flood peaks of shorter duration and a general tendency toward less perennial flow in the dry season. It also leads to a decrease in the quality of river water through storm-water drains draining polluted urban areas and, in some developing countries, sewage.

5. DEGRADATION: CHEMICAL CHANGE

5.1 Introduction

Rivers drain the landscape, and any fluxes of chemicals or changes in water balances in their catchments may ultimately be reflected in their water quality. According to Dallas *et al.* (1994) water quality can be defined as the combined effect of the physical attributes and chemical constituents of a water sample. The term 'quality' implies value or usefulness from the user's point of view, with one of the 'users' of water being the river ecosystem itself. Pro-active management of river condition requires understanding of and catering for the quality of water needed to maintain the kind of ecosystem desired.

The attributes and constituents of water vary naturally between and within rivers. Generally the quality of surface water varies throughout the year as the rate of flow varies, whilst that of groundwater is relatively stable over time as it is predominately determined by the chemical composition of the soils serving as the aquifer (Ayoade 1988; Gale & Day 1995). Under natural conditions, however, both reflect some degree of fluctuation in quality with time within some kind of long-term steady state. Human activities in the catchment and rivers affect this balance, through agricultural activities, urbanization and industrialisation by an increasing number of people (Stoffberg *et. al.* 1994). The impact on rivers is exacerbated by increasing demands for water, which result in less water being available in rivers for dilution of the pollutants draining into them (Kern 1992; Stoffberg *et. al.* 1994; Van Dijk *et. al.* 1995; Winter & Duthie 1998).

In this Chapter, variables that affect water quality are outlined in Section 5.2, types of water pollutants in Section 5.3, human activities that affect water quality in Section 5.4, and monitoring considerations in Section 5.5.

5.2 Variables that affect water quality, and their biological relevance

The reader is referred to Dallas & Day (1993) for a full description of water-quality variables and their biological significance. The following information is derived from that publication unless otherwise indicated.

5.2.1 Chemical constituents

Metals are major chemical constituents of water and most are toxic at low pH. They are defined as elements that are good conductors of electricity and whose electrical resistance is directly proportional to absolute temperature. Trace metals are those metals occurring at concentrations of one thousand parts per million or less in the earth's crust. There are two groups of trace metals: those occurring naturally in most waters and those that do not occur in measurable amounts in water. The former are mostly plant nutrients such as cobalt, copper, manganese, molybdenum and zinc. The latter, mainly cadmium, lead and mercury, are potentially toxic in water at low concentrations. Hoffmann (1995) provided a different classification, dividing the metals into macro- and micro-constituents. Macro-constituents include aluminium, copper, iron, manganese, total phosphorus, total nitrogen and zinc. The micro-constituents include arsenic, cadmium, chromium, lead and mercury.

Metals affect organisms in different ways. The majority affect enzymatic processes, or skeletal and bone functions, and are carcinogenic, while others accumulate in tissues.

Aluminium

Aluminium is one of the more toxic trace metals and is probably not an important nutrient in any organism. Its solubility is strongly dependent on the acidity of the water, being soluble at low pH levels. It adsorbs onto large organic molecules, however, and is therefore not usually available in soluble form or at toxic levels even in very acid waters.

Arsenic

Some forms of arsenic are extremely toxic. They are soluble in fats and therefore accumulate in living tissue. Although arsenic is carcinogenic in humans, many organisms have adapted to live with high levels of it. For example, algae living in lakes with high arsenic concentrations in South America are a food source for flamingos, which survive with high levels of arsenic in their tissues. Arsenic has also been shown to increase the production of root, shoot and total dry matter in the aquatic plant *Spartina patens* (Beeson *et al.* 1999).

Cadmium

Cadmium is a hazardous substance that is easily absorbed by mammals, where it becomes concentrated in tissues through a protein called metallothionein. Cadmium inhibits bone-repair mechanisms, replaces zinc in zinc-containing metalloenzymes, and is teratogenic, mutagenic and carcinogenic. When a freshwater snail species was exposed to different concentrations of cadmium, egg production ceased at 400 $\mu g\ l^{-1}$, hatching was reduced at 200 $\mu g\ l^{-1}$, and other developmental stages were affected at a range of concentrations. Beeson *et al*. (1999) reported that ingested cadmium ions adsorbed to algae caused reduced growth and reproduction in the water flea *Daphnia magna.*

Cobalt

Cobalt is an intrinsic component of Vitamin B, but is also very toxic in small amounts. Insoluble inorganic compounds of cobalt are carcinogenic when injected into mammals, while the soluble ones are toxic. They inhibit some enzymes while stimulating others.

Copper

Copper forms an essential part of enzymes in redox reactions in cells. It is toxic at low levels, but its toxicity is reduced by the presence of zinc, molybdenum and sulphate. In mammals, it is carried and bound to proteins or amino acids, and can cause brain damage.

Iron

This essential micronutrient is present in all organisms, where it plays an important role in haeme-containing respiratory pigments, catalyses, cytocromes and peroxidase. It is not easily absorbed through the gastro-intestinal tract of vertebrates, but has toxic properties at high concentrations and can then inhibit various enzymes. Iron compounds are easily oxidized, and so high concentrations of reduced forms can result in oxygen depletion in the environment.

Lead

Lead is a common and toxic trace metal. It accumulates in living tissue, and in vertebrates becomes immobilized in bone where it does not exhibit toxic effects. Lead affects organisms by interfering with the synthesis of haeme, an essential portion of the haemoglobin molecule. It also affects membrane permeability, displaces calcium at functional sites and inhibits the opening of “calcium channels” in membranes, as well as inhibiting some enzymes involved in energy metabolism.

Manganese

Manganese is an important micronutrient, at least in the glycosal transferases essential in proteoglycan synthesis in vertebrates, and possibly in other enzymes as well. High concentrations are toxic, disturbing the central nervous system and leading to disturbances in various metabolic pathways. A deficiency leads to skeletal deformities.

Mercury

Mercury is one of two metals (the other is beryllium) classified as hazardous by the United States Environmental Protection Agency (USEPA). The inorganic forms of mercury are less toxic than the organic forms. The toxic forms are stable and have long retention times in tissues. Being fat soluble, they become bound in organisms, particularly in nervous tissue, and so accumulate up the food chain. Their main effects include an increase in abnormal cell division and inhibition of enzymes. They are also genotoxic, causing chromosomal aberrations. Mercury inhibits phytoplankton photosynthesis at very low levels.

Molybdenum

Molybdenum is essential for nitrogen fixation by bacteria. It is not accumulated or toxic, except at high concentrations. It induces symptoms of copper deficiency in mammals.

Selenium

Selenium is non-metallic, but often combines with metals and acts as one. It is toxic at relatively low levels because it mimics sulphur and sometimes replaces that element in the thiol groups of some amino acids. It is an irritant in mammals, causing erosion of the joints. High rates of mortality, embryonic deformities and reproductive failures were found in water birds at the Kesterson Reservoir, California, due to high levels of selenium (Deason 1989). Selenium affects humans when they eat fish and waterfowls with elevated selenium levels (Johns & Watkins 1989), with toxic concentrations being carcinogenic and genotoxic.

Zinc

Zinc is an essential micronutrient. It forms the active site of various metalloenzymes, including DNA and RNA polymerises.

Nutrients

Inorganic phosphorus occurs most commonly as phosphate (PO_4^{3-}) while nitrogen occurs as nitrate (NO_3^-), ammonium (NH_4^+) and in many nitrogen-containing organic compounds. They naturally occur in low concentrations. Most nutrients are not toxic, even in high concentrations, but the high concentrations may have a significant effect on the structure and functioning of biotic communities. Exceptions include nitrite (NO_2^- - the intermediate form in the inter-conversion between ammonia and nitrate) and ammonia (in the free un-ionized form (NH_3), which are very toxic at higher concentrations. In the case of fish un-ionized ammonia may result in loss of equilibrium, hyperexcitability, increased breathing rate and in severe cases coma and death. Elevated levels of nutrients are usually indicative of pollution. Anthropogenic sources of nutrients may originate from point-source effluents (e.g. sewage treatment plants) or diffuse (e.g. agricultural surface runoff) or urban runoff. Nutrient enrichment, known as eutrophication, can lead to imbalances in biological communities, particularly through excessive plant growth, and associated water quality problems.

5.2.2 Physical variables

Acidity or Alkalinity (pH)

The pH of water is a measure of the concentration of hydrogen (H^+) ions in solution. Normally pH gives an indication of the acidic status (pH<7) neutral status (pH=7) or basic (alkaline) status (pH>7) of the water. The pH of aquatic systems is largely determined by geological and atmospheric influences; most freshwater streams are more or less neutral, with pH values ranging between 6 and 8.

Spot measurements of pH are almost valueless in rivers with high biological activity, as the value could change considerably during a daily cycle. However, such spot measurements can be of value in general water-quality monitoring, as long as they are taken at about the same time of day on each occasion. Changes in pH affect stream-water chemistry. For instance, as pH levels fall below neutral, metals such as mercury, lead and manganese can become highly toxic. Toxins (e.g. trace metals), nutrients and biocides can be released from sediments, and certain substances such as selenium reduced in solubility. On the other hand, at pH values higher than 8, non-metallic ions, such as ammonium, can be converted to highly toxic ammonia.

The physiological effects of changing pH on aquatic organisms can be direct or indirect. The direct effect is an alteration in the water, ionic and osmotic balance of individual organisms. The need to increase the rate of osmotic and ionic regulation places stress on organisms by increasing energy requirements, which in turn cause slow growth and reduced fecundity.

The indirect effects of changing pH values can be far more important (Bell 1971; Mulholland *et al*. 1986; Distefano *et al.* 1991; Beeson *et al*. 1998). For example, decreases in pH, or acidification, change the composition of toxins (trace metals), causing mobilization of toxic substances such as aluminium. Some species are more sensitive to low pH levels than others, but in general the effects are reduced colonization and emergence rates, decreased species diversity and density and the reduced decomposition of organic matter such as leaves. Acidity thus has a negative affect on both the quality and quantity of food in aquatic ecosystems.

Total dissolved solids, conductivity and salinity

Total dissolved solids (TDS) encompass the total quantity of dissolved material (organic, inorganic, ionized and un-ionized) in a water sample. Conductivity is a measure of the ability of a sample to conduct an electrical current, with higher conductivities reflecting higher TDS levels. Salinity is defined by the saltiness of the water, which is also determined by its TDS concentration. Conductivity is usually measured in purer waters, whilst salinity is measured in more saline waters where concentrations could be higher than field conductivity instruments can measure.

According to Dallas and Day (1993) the most commonly found inorganic ions in natural waters, also called the major ions, are the cations magnesium, sodium, calcium and potassium, and the anions sulphate, chloride, carbonate and bicarbonate. Less common inorganic ions include nitrate and phosphate and trace metals such as copper, zinc and iron while a small amount of inorganic material may also be found in un-ionized elements (e.g. hydroxides). Most dissolved organic compounds are non-ionic, however, humic and fulvic acids, also known as polyphenolics, are often present in ionic form. TDS values vary naturally in inland streams because of natural processes such as the rate and nature of rock weathering. Activities such as land clearance, sewage discharge and irrigation have artificially increased present-day levels.

Tolerance of organisms to high or low TDS levels is species specific. Very little information is available on these tolerances, but a few generalizations can be made.

- High TDS concentrations in the water surrounding an organism will draw the water out of it, causing dehydration. Toxic quantities of salts can also be absorbed by the body cells of organisms. In both cases the overall result could be a decrease in species diversity (Rutherfurd *et al*. 2000).
- The rate of change of TDS concentrations can have a greater impact on organisms than the concentration *per se.*
- Tolerance varies with the condition of the organism, the life stage, time allowed for acclimation and water temperature (Rutherfurd *et al*. 2000).
- Juvenile stages (larvae and eggs) are considerably more sensitive to extreme TDS levels than adults.
- There is a critical level of salinity for any organism, above which it cannot survive.
- High TDS levels provide some protection against heavy metals because they modify their chemical speciation.

The above findings refer largely to aquatic species from the middle or lower (foothill zone and coastal plain zone) reaches of rivers, since almost no information is available on species occupying the upper reaches. Species from the mountain-stream zone may be much more sensitive to TDS levels and changes than those further downstream because of the very pure, poorly buffered waters that they inhabit (Hart *et al*. 1990; Dallas & Day 1993; Palmer & Scherman 2000).

Total suspended solids, suspensoids and turbidity

Total suspended solids (TSS), or suspensoids, are organic and inorganic solids suspended in the water. TSS can be measured by filtering the particles from the water and weighing them. Turbidity is a measure of the degree of light penetration past the suspensoids, which directly affects visibility through the water and photosynthesis of immersed plants. Turbidity can be measured *in situ* with instruments.

Increases in TSS levels can affect various attributes of water quality, nutrient levels, the availability of habitats, and the well-being of aquatic species (Winter & Duthie 1998; Koebel *et al*. 1999; Rutherfurd *et al*. 2000). The temperature of turbid water falls because suspended particles increase the reflection of solar heat and reduce its absorption by the water, leading to a reduction in photosynthesis and thus primary production (Dallas *et al*. 1994). In shallow water, or the upper layer of stable water, high turbidity levels can have the opposite effect, because chemical alterations (e.g. low pH) may release high nutrient loads from suspensoids, resulting in the nuisance growth of aquatic macrophytes and eutrophication of the water body (Rutherfurd *et al*. 2000).

Settling suspensoids smother habitats and eggs, prevent spawning, and deposit adsorbed substances such as nutrients and trace metals. This may render adsorbed elements such as toxic trace metal ions and nutrient molecules unavailable. The former situation represents an advantage in that these toxins cannot cause harm to aquatic organisms. The latter situation, however, renders the adsorbed nutrients unavailable for plant and animal usage (Dallas *et al*. 1994).

High TSS levels increase macro-invertebrate drift (the rate at which aquatic invertebrates move by floating downstream), reducing upstream densities of benthic organisms. They increase stress levels in most fish species, reducing feeding efficiency and growth rates; and affect the food-searching ability of visually-hunting predators, as well as predator-prey interactions. They damage the gills of aquatic invertebrates and fish, and the feeding organs of filter feeders (Dallas & Day 1993).

Normally increasing TSS levels are accompanied by a reduction in species biodiversity (Dallas & Day 1993; Winter & Duthie 1998; Koebel *et al*. 1999; Rutherfurd *et al*. 2000).

Chemical oxygen demand and biological oxygen demand

Chemical oxygen demand (COD) is a measure of the oxidation of reduced chemicals in water. Biological oxygen demand (BOD) is an empirical test to determine the relative oxygen requirements for the biological degradation of organic matter (effluents, polluted waters). As organic enrichment increases, oxygen levels decrease.

Dissolved oxygen

Dissolved oxygen (DO) concentrations in rivers depend on a number of factors, such as water temperature (cold water is able to retain more DO than warmer water), and type of flow (quieter waters capture less oxygen from the air than more turbulent waters). DO also varies from river to river and diurnally, depending on the amount of the biological activities of respiration and photosynthesis. Human disturbances of rivers and their catchments have generally led to lower levels of DO in river water, through reduced flows and introduction of oxygen-demanding wastes. The wastes may be broken down chemically (COD) or decomposed by organisms (BOD).

Ecologically, low DO concentrations impact the emergence, behaviour, blood chemistry, reproduction, growth rate and food intake of aquatic organisms, and can cause death. High DO levels can be equally dangerous, with supersaturated water leading to gas-bubble disease whereby oxygen bubbles surround the gills of fish (Walden 1976; Kleynhans *et al*. 1992; Koebel *et al*. 1999; Rutherfurd *et al*. 2000).

Temperature

Water temperature influences the rate of all biological activity. All aquatic organisms, except for birds and mammals, are poikilothermic: their body temperatures are the same as that of the surrounding water. They are thus very susceptible to both increases and decreases in water temperature (Rutherfurd *et al*. 2000). Natural variations in water temperature are one of the factors dictating the balance of species in any one area, with low and high temperatures increasing or decreasing aquatic biodiversity through species either being unable to cope with changing conditions or coping better. When water temperatures are outside the tolerance range of a species, that species will be excluded from the affected area.

The solubility of dissolved oxygen in water is reduced with increasing temperatures, thus decreasing its concentration. (Arthur *et al*. 1982; Osborne & Davies 1986). Limited availability of dissolved oxygen can lead to stress, evasive behaviour or death of aquatic organisms. The same conditions also increase the vulnerability of organisms to toxic chemicals, disease and parasites (Rutherfurd *et al*. 2000). Temperature changes also affect the timing of life cycles; the development of different life stages; genetic selection; prey-predator interactions and other intra-specific interactions; and the abundance of algae and other plants (Arthur *et al*. 1982; Cravens 1999; Rutherfurd *et al*. 2000).

Unusual temperatures can cue life-history stages at inappropriate times (Pitchford & Visser 1975; Arthur *et al*. 1982; Osborne & Davies 1986; Cravens 1999; Miller 1996a; Rutherfurd *et al*. 2000). For instance, warmer water encourages aquatic insects to grow faster and emerge early, which could lead to adults emerging at times when air temperatures are not favourable or food sources are not present. Higher temperatures also positively affect the rate of photosynthesis, encouraging the growth of plants, including nuisance algal blooms.

5.2.3 Biological variables

***Chlorophyll* a**

The quantity of chlorophyll *a* contained in phytoplankton is often used to measure the rate of primary production or photosynthesis.

Counts of F. coliform, E coliform, parasites and viruses

Bacterial pathogens, viral pathogens and parasites are mostly a human or public health concern (Dallas *et al*. 1994; Hoffmann 1995). They are not addressed in this review.

Dissolved organic matter and particulate organic matter

Dissolved organic matter (DOM) and particulate organic matter (POM) occur naturally in aquatic systems, as plants and animals die and decay. They are sources of energy for aquatic life and their decomposition can lead to oxygen depletion of the water (Ayoade 1988; Dallas *et al*. 1994; Hoffmann 1995). In polluted rivers they may reach very high concentrations.

5.3 Types of water pollutants

Human disturbances to aquatic systems and their catchments pollute the waters, changing their chemical and physico-chemical properties and causing responding changes to their biological attributes. According to Miller (1996a) there are nine types of water pollutants:

- disease-causing agents (pathogens);
- oxygen-demanding wastes;
- water-soluble inorganic chemicals;
- inorganic plant nutrients;
- organic chemicals;
- sediment or suspended matter;
- water-soluble radioactive isotopes;
- thermal pollution;
- genetic pollution.

Pathogens include viruses, bacteria and parasitic worms that enter rivers from domestic sewage, and untreated human and animal wastes. Oxygen-demanding wastes consist of organic wastes, such as sewage, that can be decomposed by aerobic (oxygen demanding) bacteria. Water-quality degradation occurs as the bacteria deplete the water of DO, and can lead to the death of aquatic plants and animals. Acids, salts, and compounds of toxic metals such as mercury and lead, are examples of water-soluble inorganic chemicals. High levels of these can make water undrinkable and kill aquatic life. Inorganic plant nutrients include water-soluble nitrate and phosphates. These encourage increased growth of algae and other aquatic plants that, through their sheer numbers respiring oxygen, or through their death and decay, can deplete the water of dissolved oxygen. Fish kills are a common result of oxygen depletion through the presence of algal blooms. Oil, gasoline, plastics, pesticides and detergents are examples of organic chemicals. They can harm aquatic life and threaten the health of humans drinking the water.

TSS consists of insoluble particles of soil that become suspended in water, mostly through land erosion. These reduce light penetration and therefore photosynthesis; adsorb nutrients, toxins such as pesticides, bacteria and many other harmful substances; clog and fill stream channels and reservoirs; disrupt aquatic food webs through blanketing feeding areas; smother surfaces, reducing species abundances through covering habitats and eggs; and clog gills.

Water-soluble radioactive isotopes can cause genetic changes, cancer and birth defects. Thermal pollution results from industrial or other areas changing the temperature of stream water by, for instance, using water for cooling. The warmer water holds less dissolved oxygen, possibly influencing survival rates of aquatic organisms. Genetic pollution occurs through the deliberate or accidental introduction of non-native species into a river system (Chapter 4). These species may out compete many native species and reduce biodiversity (Miller 1996a).

5.4 Human activities that affect water quality

5.4.1 Point source pollutants

Point-source pollutants are released into rivers from easily-identified sources such as sewage-treatment works or industrial complexes. Conventional and toxic pollutants from homes, factories, and storm runoff may flow through a network of sewer pipes to waste-water treatment plants, where they undergo different levels of purification before being released into surface water bodies (Doucette *et al*. 1985; Whitehurst & Lindsey 1990; Miller 1996a). Some effluents empty directly into rivers, often with more devastating effects than occur with treated ones.

The main problems with point-source pollutants result from the following occurrences:

- direct discharge of raw sewage and other effluents;
- inadequate or no chlorination of final effluent;
- inadequate treatment of waste water due to overloading of the treatment plant by, for instance, storm water also being channeled through the plant;
- inefficient processing or poorly maintained sewer systems;
- accidental chemical /sewage spills or leakages.

Relatively few pollution problems are due to the deliberate discharge of raw sewage into rivers. Pipeline and sewer failures and industrial accidents are more common, and can exert more catastrophic effects upon aquatic systems, eradicating biological communities in the immediate vicinity of spills (Lelek & Köhler 1990; Whitehurst & Lindsey 1990; Kleynhans *et al*. 1992; Fleckseder *et al*. 1993; Dowson *et al*. 1996).

5.4.2 Non-point source pollutants

Non-point source pollutants cannot be traced to any single site of discharge. They pollute water through runoff, sub-surface flow, or deposition from the atmosphere (Pegram & Görgens 2001). The following are major sources of non-point source pollutants.

Agricultural practices

Farming activities, such as crop production and the rearing of livestock, frequently depend on the use of fertilizers, and pesticides for pest control (Shirmohammadi & Knisel 1989). These substances are sources of nutrients and toxins that ultimately drain, through direct runoff, infiltration and groundwater movement, to the river (Schulz *et al.* 2001).

Nutrient enrichment (eutrophication) of rivers can result from using fertilizers such as phosphorus and nitrogen in the catchments. The resulting proliferation of fast-growing plant species can reach pest proportions, choking streams, and creating algal blooms, scums and discolouration of the water (Shand *et al.* 1994; Quick 1995; Van Dijk *et al.* 1995; Kinniburgh *et al.* 1997; Winter & Duthie 1998; Almendinger 1999; Schulz *et al.* 2001).

Agricultural activities may also result in trace metals leaching into rivers. Deason (1989) found that concentrations of barium, molybdenum, vanadium, zinc and TDS were significantly higher at study sites in irrigated areas than at other sites. Agricultural activities in the San Joaquin River basin, California, are a major source of selenium, boron, salt and TDS in the river (Johns & Watkins 1989).

Increased soil salinity is a natural phenomenon found most often in floodplains and other low-lying areas. Dissolved salts in surface and groundwater entering these areas can concentrate there if there is restricted or no outflow. Irrigation can increase the rate of soil salinization in areas without satisfactory drainage. Salts accumulate in the soil and any subsequent irrigation with the mineralised drainage water leads to further concentration of salts on the land (Gale & Day 1995; Fogg & Wells 1998). This is increasingly becoming a problem in arid farming areas in South Africa (Davies & Day 1998).

Livestock physically alter runoff patterns to rivers, by trampling soil and plants. They also produce dung and urine, which may affect the water quality of the receiving river. Such nutrient additions through feedlots depend on the operation of the farming activity. Dispersed grazing and subsequent dung and urine production have less effect on nutrient loads reaching the river than do commercial herds kept at high densities (Miller 1996a). Faecal contamination of the river may also occur. For instance, high sheep-stocking densities in summer were thought to be responsible for the consistent high summer levels of faecal contaminants in the Forth River near Castleton in North Derbyshire, U.K. (Tranter *et al.* 1996).

Preparing the soil for crop production involves physically disturbing it. This can lead to soil erosion and subsequent sedimentation problems in the receiving river (see Forestry below). Phosphorous readily adsorbs onto clay particles. Erosion of such nutrient-rich soils can supply a significant amount of nutrients to rivers (Rutherfurd *et al.* 2000).

Biocides such as herbicides, insecticides and fungicides are used in the production of food and fibre and in the control of vector-borne diseases. Shirmohammadi & Knisel (1989) reported that the climate of the south-east and Delta states of the U.S.A. leads to high incidences of pests, weeds, diseases, insects, parasites, nematodes and fungi in intensively managed agricultural and horticultural systems. These infestations required heavy use of a wide range of pesticides, leading to pollution of the surface water and accumulation of pesticides on bottom sediments of the rivers.

Biocides concentrate in the tissue of living organisms, accumulating through the different trophic levels in the food chain in the process of bio-magnification. For instance, Sancho *et al.* (1997; cited by Chung *et al.* 1998) reported that the insecticide fenitrothion showed a strong tendency to concentrate in the brain of the European eel (*Anguilla anguilla*). Residue levels provide an indication of the persistence of biocides in water, sediments, vegetation and biota. Line *et al.* (1998) studied the persistence of atrazine residues in sub soils and aquifer sediments, reporting that it had a half-live of

1-5 years in the sub soils with no measurable degradation in aquatic sediments. Rapid breakdown occurred in the surface soils (<10 day half-life), particularly if the pH was above 7.7. Ma & Spalding (1997; cited by Chung *et al.* 1998) reported the half-life of atrazine to be approximately 223 days, whilst Bouwer (1987) found the half-lives of the more persistent chlorinated hydrocarbon pesticides in soil to be generally less than four years. DDT and Dieldrin have the longest half-lives. Soil type, rate of water movement through the soil, rainfall patterns and microbial activity in the soil all influence the persistence of biocides (Bouwer 1987; Chung *et al.* 1998).

Forestry and land management

Forestry and land management practices may include burning of vegetation during certain times of the year. Fires have been reported to volatise nutrients (Fogg & Wells 1998), and increase TDS levels in streams during the first rains after fires but with no long-term effect (Van Wyk & Lesch 1992). Scott *et al.* (1998) reported that controlled fires in three natural fynbos catchments in the western Cape had little effect on erosion rates. The annual sediment yields were variable and were dominated by exports during large storm events rather than by fires. Wildfires through nearby afforested catchments, on the other hand, increased suspended sediment loads (Scott *et al.* 1998) and erosion rates (Scott *et al.* 1998). The changes in erosion rates and sediment yields due to wildfires were related to the presence of water repellency in the soils after the fires. Water repellency led to an increased frequency and amount of overland flow over the soils, making the soils more erodible after fires. Water repellency in soils is dependant on the type of vegetation burnt. Alien vegetation such as pines contains oils, which are released during fire. Fire intensity is also greater in alien stands than in natural fynbos vegetation. The effects of fire on soils and water quality of stream water are dependant on the intensity of the fire, moisture contents and wetness of the catchment, the presence of a riparian zone and how fully it was developed. It further depends on percentage riparian zone burnt, type and recovery rate of the vegetation, and the time when the fire occurred before the onset of the rainy season (Scott *et al.* 1998; Van Wyk & Lesch 1992).

Mining

The exploration, extraction, processing and transportation of coal, minerals, sand, gravel and other materials have had and continue to have profound effects on rivers. Surface and subsurface mining in the catchment, including strip mining, open-pit operations, dredging, placer mining and hydraulic mining, can affect the river corridor (Fogg & Wells 1998).

Mutz (1998) studied the effects of strip mining on the landscape, topography and stream systems in the Lower Lousitia, eastern Germany. Intensive extraction of brown coal (lignite) since the start of this century has led to sands with sulphur contents of 4-5% and little acid-neutralization capacity being re-deposited at the soil surface. The remaining lakes and streams fed by groundwater from these areas have become acid, with pH values of less than 3. The re-deposited sand undergoes a chemical reaction (sulphide oxidation) when exposed to water and air, which releases iron and toxic metals and causes the pH change.

This chemical reaction, and the subsequent leaching of harmful substances, is the basis of the problem of acid mine-drainage (Dallas & Day 1993; Gray 1996; Fogg & Wells 1998). In general, metals released by acid mine-drainage depend on the geological character of the host rock (Gray 1996). The

Hocking River, Ohio, U.S.A., for instance, has a long history of degradation due to acid-mine drainage from abandoned mines. Recorded pH levels were less then 4.5, with high dissolved concentrations of Al, Fe, and Mn (Fogg & Wells 1998). West Squaw Creek, California, U.S.A., on the other hand, received large quantities of sulphuric acid and heavy metals, such as iron, copper and zinc (Fillipek *et al.* 1987).

Drainage of such contaminated water into streams can profoundly impact the aquatic ecosystems. The fauna of such streams are usually impoverished, consisting of widespread, hardy species resistant to acid conditions. Some species can take advantage of these conditions however. Fauna found in naturally acidic Western Cape streams, for example, have colonized acid-polluted Gauteng streams (Dallas & Day 1993).

Mining can also affect the quality of river sediments. Trace elements adsorb to river sediments, with fine-grained sediments generally adsorbing more than coarser ones. Benthic organisms, near the base of the food chain, ingest organic material with its associated sediments and trace metals, with the metals then accumulating through the food chain. For example, Deacon & Driver (1999) studied streambed–sediment samples from the Southern Rocky Mountains where gold was mined for decades. They found that concentrations of arsenic, cadmium, copper, lead and zinc were generally orders of magnitude higher at river sites in mining areas than those in non-mining areas. Colour of the stream waters in the old gold-mining areas in the Rockies ranges from normal to orange, bright yellow or powdery white, and many are completely devoid of life (J. King, pers. comm.). In the Sacramento River basin, mercury used for recovering gold from deposits caused additional water-quality problems and contamination of sediments (Roth *et al.* 2001).

Runoff from roads

There are many potential sources of contaminants in the runoff from road surfaces and rooftops, or through fallout from industries and car exhausts. Almost all pollutants deposited on impervious surfaces, if not removed by street cleaning, wind or decay, will eventually drain toward a river or other aquatic system. Potential pollutants include (Dallas & Day 1993; Ellis *et al.* 1997; Armitage *et al.* 1998):

- degrading road materials;
- vehicle parts (fuel, lubricants, hydraulic fluids, coolants, lining, exhaust emissions, rust, vehicle components);
- atmospheric fallout (from industrial stacks and vents);
- litter (packaging (plastic, paper, tins and bottles), food discards, animals (dead animals and droppings), vegetation (branches, leaves, twigs, rotten fruit and vegetables, grasses);
- spills (sand, gravel, cement, and agricultural, chemical and petroleum products);
- de-icers applied to roads, airports and aircraft to prevent ice formation and build-up;
- construction material (concrete, shutters, planks, timber props and broken bricks);
- domestic pesticide sprays and herbicides used to control roadside weeds;
- unauthorised dumping or washing (chemicals such as detergents and oils);
- miscellaneous (e.g. old clothing, shoes, pens, balls, cigarette butts)

Impacts on aquatic ecosystems will depend on the type of pollutant. Ellis *et al.* (1997) showed that a range of hydrocarbons derived from lubricating oils led to high mortalities in the isopod *Asellus* *aquaticus* and the snail *Lymnaea peregra*. Also, runoff from roads impacts negatively on the primary food source of rivers. The food quality and temporal availability of particulate organic matter were significantly less in streams receiving runoff from roads compared to streams not receiving runoff (Ellis *et al.* 1997). This can lead to a reduction in species diversity since only species adapted to the changed food source will be able to survive.

5.5 Monitoring and biomonitoring

Most countries have comprehensive networks for monitoring the chemical constituents and physical attributes of their rivers' waters. Until recently, these provided the main information on the health of a nation's rivers. Three main shortcomings have emerged from this practice. First, the monitoring data were usually derived from spot samples of water taken from the river at regular (daily, weekly or monthly) intervals. These provided no information on, for instance, the occurrence of polluting incidents outside the time of sampling. Second, although analyses of the water gave great detail on its chemical and physico-chemical properties, there was virtually no understanding on the implications of this for the aquatic plants and animals (Dallas & Day 1993; Dallas *et al*. 1994). Third, much of the degradation of rivers has been due to destruction of physical habitat or the natural flow regime, and not to chemical pollutants.

These shortcomings have led to the development of biomonitoring techniques, or the use of plants and animals, to monitor the health of rivers. The biota integrate the cumulative effects of the chemical and physical environment, with the presence or absence of different species being indicative of prevailing conditions in the river over an extended period of time. Presence and absence data, and biological indicators such as behaviour, biomass, biodiversity, or rate of photosynthesis or respiration, can thus all be employed to give an indication of ecosystem "health" or integrity. However there are considerable disadvantages involved in using biological communities for water quality assessments (Roux *et al*. 1993; Dallas *et al*. 1994; Dallas & Day 1993; Copeman 1997; McQuaid & Norfleet 1999). Dallas & Day (1993) listed these as follows:

- quantitative sampling is generally problematic and requires considerable sample replication;
- data processing and analysis are often labour intensive and time consuming;
- the uncertainty of the taxonomic status of certain species often requires the input of specialists;
- biological communities are not sensitive to all pollutants and their responses to toxins have not been well documented;
- factors other than water quality variables may effect these aquatic organisms and should thus be taken into account when drawing any conclusions;
- it is very difficult to distinguish if observed changes are natural or the result of pollution.

Biomonitoring has taken on new prominence in South Africa since passing of the 1998 Water Act. In this Act, aquatic ecosystems are recognised as the base of the resource, and it is required that their health be actively managed in order to protect the benefits that they provide humans. Biomonitoring is still a science in its infancy, with much development of general principles and concepts to be done. It is becoming integrated with chemical and physical monitoring, to provide a new holistic approach of many interacting components for assessing aquatic ecosystem health (Dallas *et al*. 1994). As such, it provides a promising tool for restoration projects.

5.6 Summary

Water quality varies between rivers due to climate and the underlying geological formations, and through the year due to seasonal changes in flow and temperature and responding biological activity. Humans impact on water quality and quantity through changing the drainage patterns to rivers and the chemical constituents and sediments the water may be carrying. They also use rivers as wastewater conduits and feed storm-water drains that have drained dirty urban areas into them. Pollutants that can occur in rivers because of this include disease-causing agents, oxygen-demanding wastes, water-soluble inorganic chemicals, inorganic plant nutrients, organic chemicals, sediment or suspended matter, water-soluble radioactive isotopes and thermal pollutants. These generally impact aquatic ecosystems by reducing abundances and biodiversity, and their general ability to function efficiently.

Rehabilitation of degraded systems requires that attention be paid to the quality of the water, as well as the quantity (Chapter 4) and the physical attributes of the channel (Chapter 3). Unless all three are suitable, planned rehabilitation may be unsuccessful.

6. PLANNING REHABILITATION

6.1 Introduction

It is difficult to combine all aspects of river rehabilitation to give a "holistic" view (Bren 1993). Holism is defined as: "…the tendency in nature to form wholes that are more than the sum of the parts by creative evolution…" (Concise Oxford Dictionary 1997). The concept of cumulative effects provides the chance to develop a holistic view, but this implies that an understanding of the whole river system needs to be in place before a river rehabilitation project can be started. Acquiring such information on the linkages between separate ecosystem components is a formidable challenge (Bren 1993).

This Chapter outlines the agents involved in an holistic rehabilitation project (Section 6.2), the concept of Integrated Catchment Management (Section 6.3) and Inter-disciplinary planning and design (Section 6.4).

6.2 Major participants in river rehabilitation

The major participants or 'interested parties' can be summarised as: the authorities responsible for managing the river; the specialists who will provide the information upon which decisions will be based; the landowners who will be affected by the rehabilitation process; and the general public.

Henry *et al.* (1995) listed the participants in a "restoration" project on the Rhône River, Geneva. It included: managers of the project; scientists and environmental consultants on sediments, aquatic vegetation, invertebrates and fish; river engineers; socio-economic participants; local landowners including the municipality; Non-governmental Organisations; government administrations; Rhône River management authorities; water and navigation agencies; businesses involved in woodcutting and dredging; and the financing agency. Sometimes the term "community" is used to describe the participants in a river rehabilitation project. Kern's (1992a) community for rehabilitation of the Upper Danube and Rhine Rivers consisted of administrative, nature conservation and water authorities; farmers' representatives; and fisheries experts. The community for the Murrumbidgee Catchment Action Plan for Integrated Natural Resource Management in Australia consisted of the local government; the New South Wales state government agencies; the Australian Capital Territory (ACT) state government agencies; the commonwealth government; environmental groups; industry representatives; educational institutions; research institutions; utilities representatives; business groups and professional associations such as engineers (Shepheard *et al.* 1999). The major players in such rehabilitation communities are further discussed below.

6.2.1 Management

Initiation and management of river rehabilitation schemes usually rest with local authorities and government agencies (Kern 1992a; Shepheard *et al.* 1999). Besides their responsibilities for planning and guiding projects, they also fund and subsequently assess how money is used. If local funding is not available, funds from sources outside the local government may be sought. The Australians, for example, make use of funding programmes such as the National Heritage Trust or Land Care programmes. Management can also target actions that attract funding from outside sources such as the business sector (Shepheard *et al.* 1999).

6.2.2 Specialists

The specialists involved depend on the specific requirements of the project (Kern 1992a). Civil, hydraulic and flood-defence engineers may be the first to be involved as are, more recently, ecologists (Brookes 1990; Kern 1992a; Holmes 1993; Large *et al*. 1993; Sear *et al*. 1995; Nolan & Guthrie 1998). Holmes (1993) pointed out that effective communication between engineers and ecologists is vital. The ecologists, including botanists and fish and invertebrate biologists, should understand why engineering work is needed, while the engineers should understand why certain river features are being recommended for safeguard or creation. Fluvial geomorphologists have also begun to make important contributions, describing the environmental problems ensuing from interventions to river channels and subsequent adjustments of the river (Brookes 1987). Other specialists who may be used include palaeoecologists and historical geographers (Large *et al.* 1993), experts in pollution control and water resources (Nolan & Guthrie 1998), and hydrologists.

Inclusion of such experts as a co-operating team throughout the project imparts a comprehensive and scientific basis to it. Ensuring that the experts employed are those with the best available knowledge will increase effectiveness of the project, hopefully minimising inappropriate works (Lucas *et al.* 1999).

6.2.3 Riparian landowners

Riparian landowners are major affected parties in rehabilitation projects, especially where these take place on privately owned land. River rehabilitation can only become a reality if riparian landowners are willing to sell their land, or are compensated in some other way for reduced usage and crop losses (Cals *et al.* 1998; Piégay *et al.* 1997; Vivash *et al.* 1998). Careful planning with landowners is critical to the success of the project.

Melbourne Water, for example, undertook rehabilitation projects such as fencing and re-vegetation on privately owned land along the Marybyrnong River, Australia, in co-operation with the landowners. The landowners signed agreements to be responsible for maintenance once the works were completed. Funding and technical support was made available for this, in an open and accountable manner, after one-on-one negotiations between the authorities and the landowners. This led to support by the landowners for the rehabilitation work and confidence in it (Fisher *et al.* 1999). Without such co-operation, rehabilitation projects can be seriously hampered or stalled. Because of this, projects are often planned for areas involving only one landowner, thus simplifying negotiations (Eden *et al.* 1999).

Denial by farmers that agricultural activities can impact the environment is a confounding problem. Farmers in the Upper Murray catchment thought their activities had less impact than sewerage effluents discharging into urban reaches (Terril 1999). Some believed that stock did not cause environmental damage, as only a few areas were used to access the stream and so their overall impact was limited and negligible. Others felt that fencing was not environmentally sound, since control of weeds and vermin was difficult along fences. The cost of maintaining fences was also a problem, especially in flood-prone areas where fencing cannot be along a straight line. These and other costs, such as for re-vegetation, de-motivated farmers and stopped them from participating in rehabilitation projects. The negative attitude of government officials towards farmers was also a problem, with farmers feeling that government should not impose its will but rather partake in consultation and negotiation.

Consultation with riparian landowners usually follows the same process as that of public consultation.

6.2.4 General public

Due to differences in terminology it is often difficult to distinguish the various participants in a project. For example, Shepheard *et al.* (1999) used the term 'community' for all the participants in a case study. Sometimes, 'community' appears to refer to the local riparian people who will be affected by a project. The term "general public" used in this literature review refers to the people dependant on the river for domestic and recreational use (the so-called 'Interested and Affected Parties'), as well as Non-Governmental Organisations (NGOs) who have an interest in environmental issues but are not directly affected by rehabilitation.

Consultation with the general public is becoming a prerequisite for river rehabilitation projects. South African law requires that proper public consultation be held before the onset of a project, as outlined in the Environmental Conservation Act of 1989 (Barnard 1999). This process usually consists of a series of public meetings, advertisements in local newspapers, and the use of displays in public areas (Luger 1994; Vivash *et al.* 1998; Lucas *et al.* 1999; Travers & Egbars 1999). In Australia, pamphlets are distributed to relevant households (Travers & Egbars 1999). The outcomes of meetings are published in local newspapers and are available at local libraries. For rehabilitation projects on the River Brede in Denmark and the Rivers Cole and Skerne in the U.K., a community liaison officer was appointed to create and maintain ongoing links with the general public throughout the project (Vivash *et al.* 1998).

Such avenues for voicing public concern and comments help locate and rectify local problems. Residents of the Sandvlei area near Macassar, Western Cape, for example, showed that several houses were regularly flooded, and that *Typha* was invading pastures due to an elevation in the water table (Luger 1994). As a result, the area was included in management plans for the Kuils River, with specific recommendations drafted to alleviate the problems. In a project in the Eastern Cape, where the aim was to use people-centred approaches for riparian zone rehabilitation, the community collectively identified the need to investigate sedimentation at a causeway (Motteux *et al.* 1999). This led to ordinary people realising their power to take action and address their problems. The exercise created avenues of communication between scientists and the villagers, with all involved becoming equal partners (Motteux *et al.* 1999).

The public consultation process can also be used to investigate peoples' perceptions of their environment. Tunstall *et al.* (1999) studied the public's response to and perception of a rehabilitation scheme on the River Skerne, U.K., before and after completion of the project. The study showed a positive response to the project, as the public had been informed and involved throughout and so shared the enthusiasm of the managers and scientists. River perception studies reveal a trend of people in urban areas tending to prefer river rehabilitation whilst people in rural areas tend to favour flood defence channels to protect their agricultural practices (Downs & Thorne 1998).

In general, peoples' perceptions towards the natural environment seem to be highly dependent on their socio-economic status and level of education. Poorer people may have little or no regard for their environment because of other necessary priorities, with rivers used as areas for disposing of wastes or washing clothes. On the other hand, middle and upper class income groups may have more regard for their environment. In all groups, social status may also influence the level of involvement in rehabilitation. Tunstall *et al*. (1999) called for research to be done into public responses to projects on different kinds of rivers in different settings and landscapes and in different countries and cultures.

Public training and awareness campaigns help people to understand how to care for their environment, and create a sense of belonging. "People can help sustain if they know how" (Manson & Darlington 1999). By enhancing the public's understanding of catchment processes and the complexities of river systems, an understanding of how to manage these can be reached (Boyd *et al.* 1999; Travers & Egbars 1999). The community education/marketing strategy formulated for Cristie Creek catchment, Australia was called "Water Catchment Care" (Manson & Darlington 1999). Awareness of the catchment was created through displays of maps and three-dimensional models. Major traffic networks, schools and shopping centres were highlighted, to enhance and expand the peoples' perception of where they live. The public awareness campaign in Cristie Creek catchment revealed a interest in catchment care management/education, which could be exploited at a national level in Australia (Manson & Darlington 1999).

Versfeld (1995) gave an example of the extent to which the general public can be involved in the planning process. In a study in the Stoffelton catchment in KwaZulu/Natal, Participatory Rural Appraisal (PRA) was applied with a freehold rural community. The people, who were knowledgeable about catchment processes, identified a number of issues relating to catchment rehabilitation in their environment, thereby contributing to the plans. They then sought continued involvement after the planning phase, fearing that "outsiders" or "experts" might impose a top-down and prescriptive mode of management. Questions raised by the project included: how could local knowledge be captured; to what extent should the public participate during the implementation phase; and what was the sustainability of the agreed plan? The study revealed a real need for people-driven land rehabilitation, and that PRA has application wherever people are involved in or affected by decision-making (Versfeld 1995).

The New South Wales Department of Land and Water Conservation in Australia involved the general public in stream assessment, stream management plans, planting of trees and the construction of erosion control structures for their waterways. Involving the general public imparted a feeling of ownership towards the project (Smith 1999). In a government-locals partnership, a feeling of ownership was created, the project was supported, and environmental knowledge and beliefs of the locals helped developers in the design of the project.

6.3 Integrated Catchment Management and Adaptive Management

The concept of Integrated Catchment Management (ICM) is two-fold. The first part addresses the physical interaction of catchment processes that result in the different kinds of rivers. The second part addresses how management is based on an understanding of these processes and effectively encompasses the a whole catchment. ICM brings together all participants in a management or rehabilitation project, promoting cooperation between the various disciplines (Leentvaar 1997).

6.3.1 Understanding catchment processes

"River systems are complex hydrological and ecological continua extending from the catchment to the sea. They comprise not only flows of water, but also flows of sediment and energy." (Harper & Everard 1998). A clear understanding of these interacting catchment processes is vital in river rehabilitation work, with Large *et al.* (1993) stating that areas beyond the riparian zone must be considered when conserving and rehabilitating river corridors. These include floodplains, wetlands, seeps, and the general topography and geology of the landscape. Reactive surveys are often limited to investigating degraded river reaches, but a lack of attention to upstream and downstream reaches can lead to unawareness of potential indirect adverse impacts (Holmes 1993). These can include changed sediment and flow regimes in these areas.

6.3.2 Management within an Integrated Catchment Management framework

Authorities responsible for managing and initiating rehabilitation projects should understand interacting catchment processes in order to effectively manage river ecosystems. Harper & Everard (1998) stated that dynamic and intimate interactions between river channel, riparian zone and adjacent floodplains are generally overlooked by managers. Decisions may be made on a small-scale, short-term, site-by-site basis, when it would be more appropriate to address long-term sustainability of all aquatic resources in the landscape. For sustainability, large-scale and comprehensive rehabilitation projects are preferable because they are more likely to lead to a self-sustaining river ecosystem. An integrated approach addressing the major ecological interactions within the catchment is also more likely to succeed than attempts to manage for a single species (Brookes & Shields 1996). Holistic catchment planning as an approach to river management is becoming stronger in England and Wales (Holmes 1993), and in Australia (Boyd *et al.* 1999; Manson & Darlington 1999; Shepheard *et al.* 1999).

6.3.3 Adaptive Management

Adaptive management refers to monitoring the outcomes of management, and the consequent refinement of management activities to improve the outcomes. Hypotheses about the effects of management strategies can be tested through application just as ecological hypotheses are tested through scientific investigation. Monitoring the effects of management allows results and learning to be incorporated into the next cycle of management decisions (Mitchell 2000). Adaptive management is thus a way of managing a system of which there is high uncertainty, and can be used to provide new understanding. For instance, in fisheries management, fish populations can be deliberately over-estimated or under-estimated to examine the population's response to harvest pressures (Williams 1998).

6.4 Inter-disciplinary planning and design

A structured planning and design procedure with community involvement is crucial for successful river rehabilitation (Versfeld 1995; Cals *et al.* 1998; Rutherfurd *et al.* 2000). Without it, "band-aid solutions with inefficient outcomes" may result (Kapitzke 1999). Many rehabilitation projects lack a clear planning procedure and thus fail to recognize key steps (Kapitzke 1999; Koehn *et al.* undated). The nine-step planning and design procedure presented below is a combination of different steps used by different authors. Each step has strong linkages to other steps and the planning procedure as a whole. Managers can return to earlier steps and reassess them in the light of later steps, and so feedback between steps is important.

6.4.1 Step 1: Developing goals and measurable objectives

Setting goals and objectives is a necessary first step, in order to give the project direction and commitments. According to Holmes (1993) rehabilitation goals should be defined from the start so that everybody understands the aims and knows when and whether they have been reached. It is essential to identify realistic goals and objectives (Nilsson 1996; Ladson *et al.* 1999), as unrealistic ones create expectations that cannot be met (Fogg & Wells 1998). These objectives and goals should be clearly stated, not only qualitatively but also quantitatively where possible (Kondolf 1995a). Good goals can be long-term or short-term, and will have some sense of an endpoint or clear target. They direct the project and are thus based on a statement of intent or purpose. A wide range of goals is possible for river rehabilitation projects, ranging from sustaining stable and diverse aquatic ecosystems to restoring natural processes in a river (Nilsson 1996; Rutherfurd *et al.* 2000).

Objectives have the following attributes.

- They are more site specific than goals.
- They give direction to the general approach, design and implementation of the rehabilitation effort.
- They support the goals and flow directly from the identification and analysis of problems and opportunities.
- They should be stated with sufficient precision to be a basis for the selection of variables to be measured in the evaluation phase (Fogg & Wells 1998).
- They provide targets against which to measure success.

Setting goals and objectives requires trade-offs to be made between aspirations for the aquatic ecosystem and the realities of the situation. Scale needs to be addressed, recognizing that rehabilitation cannot be implemented in isolation but must be done in the context of the surrounding landscape (Fogg & Wells 1998). When goals are not stated, vague or inconsistent ones may evolve during the selection of rehabilitation techniques and study designs, and their future evaluation will be very difficult (Ladson *et al.* 1999; Rutherfurd *et al.* 2000).

It should be noted that in other scientific fields the hierarchical levels of goals and objectives may be the reverse of those indicated here. In the Kruger National Park Rivers Research Programme (KNPRRP), for instance, the objectives provide overall vision to direct management actions (Rogers & Biggs 1999). In the KNPRRP objectives hierarchy, the goals form the lowest level and define measurables that can be used to assess the efficacy of management actions.

6.4.2 Step 2: Pre-project analysis of river condition

Understanding the past and present nature of a river is an essential prerequisite for its wise management. The following three aspects provide essential background to any rehabilitative effort.

Reference condition

The natural undisturbed condition of the river is often used as a template for rehabilitation activities. This condition may well be unknown or poorly known and have to be hypothesised from any available sources of information. Possible information sources include: aerial photographs, cadastral maps, published and unpublished works, field notes accompanied by photographs, written reports, archival sources on its past nature, gauging records, boundary lines, relic material, surveys for bridge and pipeline crossings, private papers, generic models of healthy rivers and study of remnants of the system that are still in good condition today. The reference condition places the current condition of the river into context and helps development of realistic rehabilitation goals and objectives (Rutherfurd *et al.* 2000). Without such information, rehabilitation projects are often based on assumptions about what the river could be like, from anecdotal information and theoretical analysis, or local aspirations, often leading to expectations that have a limited chance of achievement (Trexler 1995; Starr 1999). It would be inappropriate, for instance, to create cobble riffles in a sandy-bed lower river, as riffles do not naturally occur in such reaches and could disappear within one or a few years due to becoming covered in fine sediments.

Present condition

Analysis of present condition provides a measure of how far removed the river is from its natural condition. The analysis is often based on a rapid assessment of the river ecosystem, done by one or a team of experienced river scientists. Chapter 7 provides further details.

Baseline data

A comprehensive rehabilitation project will usually require more detailed information on the present condition of the river than can be provided by a rapid assessment. The purpose of the baseline data is to place the rehabilitation site in context within its catchment and river zone, and to provide detailed data on key river attributes of importance for the rehabilitation effort. The selection of appropriate attributes and variables for measurement should follow logically from the project objectives. Data collection and analysis are often time consuming and costly, and may prove useless if the data collected do not allow evaluation of the rehabilitation work (Kondolf 1995a). According to Kern (1992a) there are normally five broad topics covered when gathering baseline data:

- hydrological and hydraulic data;
- channel morphology and stream type;
- ecological habitats;
- limnological investigations;
- historic, present and future land use.

Once baseline data have been collected and analyzed, the development of a rehabilitation plan and procedures for implementation and evaluation can commence. A process of thorough and clear project documentation should be included from the start (Kondolf & Micheli 1995; Large & Petts 1996; Starr 1999).

6.4.3 Step 3: Prioritization and project constraints

Cost-effective rehabilitation requires priorities to be set regarding which issues or river attributes should be addressed first (Lucas *et al*. 1999). In the past, stream management concentrated on priorities such as controlling erosion in streams, floods or pollution. Modern ecological rehabilitation of rivers demands a different set of priorities, focusing on identifying desirable "natural" features of rivers that need protection and enhancement. At some point, these have to be integrated or combined with utilitarian (e.g. flood control and erosion) priorities, usually as part of a political decision-making process that reflects the values of the people involved (Rutherfurd *et al.* 2000). Rutherfurd *et al.* (1999) stated that rehabilitation priorities should be based on conserving the highest possible level of natural biodiversity. Biodiversity can be defined as the sum of all native species of living organisms, the genetic variation within them, their habitats and the ecosystems of which they are an integral part (Lucas *et al.* 1999). Rutherfurd *et al.* (2000) believed it is more efficient to conserve entire reaches then to "attempt to rescue individual species and leave reaches to be destroyed". They identified the following principles for setting stream rehabilitation priorities:

- save what is good or valuable before trying to fix what is bad or common;
- where possible, set priorities from large to small scale, from national to reach scale;
- work on reaches and problems in the following order: rare reaches before common ones; reaches in good condition before those in bad condition; deteriorating reaches before stable or improving ones; and reaches with easy rehabilitative solutions before those with complex ones;
- spend less time on rehabilitating severely degraded reaches, and focus effort on reaches where success is more likely;
- identify the most important problems and first attend to key problems that affect all else;
- identify links between problems within a reach.

Financial constraints often limit or prevent the development of a comprehensive rehabilitation programme. Frequently, a budget for post-project evaluation is not included in the original cost estimates and, if included, is often diverted to seemingly more pressing needs. White *et al*. (1999) stated that river rehabilitation often involves additional costs such as for implementing or modifying structures to prevent damage to river health. An example of the latter includes rock chutes designed at a slope of around 20:1 to allow fish passage. Pre-project assessment, post-project evaluation and maintenance are also included in these additional costs (White *et al*. 1999). Continual re-evaluation of the relationship between available resources and plans should ensure that functional and financial constraints do not prevent the aims of the project from being achieved (Koehn *et al.* undated; Kondolf & Micheli 1995; Vivash *et al.* 1998).

6.4.4 Step 4: Strategies for rehabilitation

The chosen rehabilitation strategy should be directly related to the objectives and goals. If Steps 1 and 3 have not been addressed adequately, selection of an appropriate strategy will be compromised. Where there is good understanding of the goals and the constraints to achieving them, strategy development is less likely to be a problem (Ladson *et al.* 1999). Basic strategies include, for instance, stream stabilization, river rehabilitation, non-intervention, flood mitigation and catchment management.

Without adequate understanding, managers or project planners are likely to select techniques and treatments with which they are familiar and discard a strategy that seems too difficult to implement, even though it may be the best, or even the only, strategy that can ensure a satisfactory outcome. Any strategy should also consider humans as a part of aquatic ecosystems and the rehabilitation process, and perhaps include strategies for changing human behaviour (Wissmar & Beschta 1998; Rutherfurd *et al.* 2000). Central to developing strategies is the recognition that ecological rehabilitation may not always be possible or successful. Some river rehabilitation strategies can lead to a dramatic initial improvement, and then a decline with time. River rehabilitation is only successful if river condition remains improved decades after implementation (Wissmar & Beschta 1998; Rutherfurd *et al.* 2000). An example of a successful strategy involving channel stabilization was reported by Haltiner *et al.* (1996) and Kondolf (1996). The Schulte Road Project consisted of a 10 km reach upstream of the Carmel River mouth, southeast of Monterey, California. Due to the extraction of groundwater and lowering of the local water table, bank vegetation was lost and the channel became destabilized. The purpose of the project was to stabilize the channel again. Works, started in the 1980s, included the construction of a pilot channel and floodplain of appropriate dimensions, planting of riparian vegetation on banks (e.g. willow mattresses) and floodplain, drip irrigation, and post-and-wire reinforcements on outside bends. Data from repeat surveys of cross sections and long profiles following the 1992, 1993 and 1995 flooding events were compared with similar earlier data. Overall the project successfully re-established riparian vegetation, improved channel stability, created riparian bank habitat and improved aquatic habitat by deepening pools and creating overhanging vegetation (Haltiner *et al.*1996; Kondolf 1996).

6.4.5 Step 5: Study design, techniques and rehabilitation alternatives

This stage is concerned with the design of the structures that will be built or the actions that will be taken. Designed structures should enhance the stream's natural recovery, and have long-lasting effects even if the structures ultimately rot away. Contingency measures should be in place for implementation if the structure fails to meet the objectives (Kondolf & Micheli 1995; Rutherfurd *et al.* 2000). All the factors (direct and indirect) that influence a reach should be studied, understood and explicitly accounted for in the design process. If they are not, the forces that degraded the reach in the first place are likely to do the same to the rehabilitated reach. If possible, pilot studies should be employed to assess the merits of different designs (Rutherfurd *et al.* 2000).

Evaluation techniques should be reproducible and generate the most meaningful information possible at the least cost. An example of a commonly used technique involves using the same permanent transects for the evaluation of a number of characteristics (e.g. geomorphological features, sediment dynamics and vegetation distributions). Reproducibility preserves the integrity of data collected and permits comparison with other projects. By the end of this step the relationship between objectives, rehabilitation measures, evaluation success criteria, contingency measures and evaluation techniques should be fully explored and defined (Kondolf & Micheli 1995; Rutherfurd *et al.* 2000). As river rehabilitation project plans seldom include all the above-mentioned steps (steps 6.4.1 to 6.4.5), a hypothetical example by Rutherfurd *et al.* (2000) was used in the following illustration.

A broad goal for managing Mythic Creek, Australia, was drafted in step 6.4.1. The goal stated that Mythic Creek will be rehabilitated so that the plants and animals that occupied the area in the past may return, and natural channel processes (e.g. flooding) be restored. In step 6.4.2 pre-project analysis involved dividing the creek into five reaches. Reach 1a was a granitic North Branch which had been partially cleared; Reach 1b was a sedimentary South Branch which had been cleared extensively. Reach 2 was from the confluence of both Branches to the beginning of the Gorge section. Reach 3 was the Gorge section while Reach 4 was from the downstream end of the Gorge to a tributary confluence downstream. The latter was a channelised stream. A template of what the river would have been like was constructed with reference to historical studies and local landowners. Reach 1a appeared to be close to the template condition. The overall problems identified in the creek included degraded riparian vegetation (all reaches), poor water quality (all reaches), low biodiversity and habitat complexity (all reaches), erosion problems (reach 2), alien invasion (reach 2), cattle trampling (reach 1a), impacts from proposed dam (reach 3-4) and the lack of channel-floodplain connectivity (reach 4). Assets were reach specific. In step 3 (prioritization and project constraints) the rehabilitation priorities for Reach 1a, for example, were a proposed dam, stock grazing, and woody weeds and vines. Step 4 (strategies for rehabilitation) included the following possible approaches for Reach 1a: prevention of the dam being built, control of weeds and revegetation. These strategies are concepts of how to solve the problems, and not meant to include a detailed rehabilitation design.

6.4.6 Step 6: Feasibility of the study

In this step, the feasibility of the objectives, the strategy, the design and the techniques is reviewed. Feasibility can be assessed in terms of the following (Rutherfurd *et al.* 2000):

- costs (implementation, post project and other);
- legality (legislative or administrative constraints);
- the advantages and disadvantages;
- potential to resolve the problem;
- potential for modification if needed.

If the project is acceptable in each of the above categories, then it can be implemented. If it is found wanting in any category then re-assessment would be appropriate. Three possible options for this are:

- modify through compromise;
- employ small-scale trial runs or pilot studies;
- if the problem cannot be resolved, terminal unfeasibility must be recognized and planners would then move on to the next priority problem.

At the end of this step it will be resolved which part(s) of the rehabilitation plan will actually be implemented (Koebel 1995; Rutherfurd *et al.* 2000).

6.4.7 Step 7: Implementation: construction plan and clean-up phases

Implementation consists of all the activities needed to execute the design and achieve the project goals and objectives. An implementation plan guides the project in the following ways: the start of each task is scheduled; each step can be budgeted for; and work in each area flows logically (Fogg & Wells 1998; Rutherfurd *et al.* 2000). There are five components to implementation.

- Ensuring funds are available (Rutherfurd *et al.* 2000).
- Identifying the tools for the job. Generally, these would be ones that least disrupt the river (Brookes 1990).
- Assigning the responsibilities of implementation among the participants, and securing commitments from them. This process is important, and often overlooked (Fogg & Wells 1998; Rutherfurd *et al.* 2000).
- Installing the rehabilitation measures in accordance with the project design.
- Post-operation clearing up (Brookes 1990).

Installation involves several stages, including site preparation, site clearance, site construction and site inspection. The clean-up phase generally involves replanting or reseeding banks and exposed floodplain areas with native vegetation. Stockpiles of useful materials should be retained at strategic locations along the rehabilitated reach, while spoil that is removed can be utilized to correct drainage conditions in floodplains (Brookes 1990). In general, the implementation plans should always be kept up to date and be focused on the long-term.

6.4.8 Step 8: Monitoring and maintenance during the implementation phase

Monitoring, evaluation and adaptive management are central to the success of the rehabilitation effort (Ladson *et al.* 1999). Monitoring should be guided by predetermined criteria and allow for the recording of results in regular monitoring reports. There are three kinds of monitoring.

- Pre-rehabilitation monitoring.
- Monitoring during the actual implementation of the project. The purpose of this is to ensure that the rehabilitation plans are carried out correctly, and that any natural areas surrounding the site are protected (Fogg & Wells 1998).
- Post rehabilitation monitoring (Brookes 1990). This is addressed in Step 9.

All three types of monitoring are needed for rehabilitation success. Results in a monitoring report inform stakeholders of the progress made towards the intended goals.

The goals of a monitoring plan include the following: assessment of the rehabilitation effort relative to the project goals; provision of information that can be used to improve rehabilitation performance; provision of general information about the rehabilitation project; indicating acceptable and unacceptable results; and the provision of a budget to cover the costs of remediation options or objectives and deficiencies or unacceptable results that might follow from them (Fogg & Wells 1998).

Running in parallel with the monitoring activities should be a formal maintenance plan that includes provision for routine site inspections. A range of activities needing careful planning and budgeting might be included in a maintenance plan, such as the removal of accumulated debris (Brookes 1990).

6.4.9 Step 9: Project review: Post-project assessment

The evaluation of stream rehabilitation projects is necessary in order to determine whether these are achieving the goals and objectives that were identified early in the process. The terms evaluation, appraisal and auditing are usually used as synonyms. According to Rutherfurd *et al.* (2000) evaluation is defined as an assessment of the effectiveness of a strategy. It is normally based on monitoring results and differs from monitoring in that it involves an assessment of the success or failure of the strategy, not just its description. Despite the importance of evaluating the success of rehabilitation projects, this is more often than not done poorly or not at all. In some cases post-project evaluation is simply omitted, while in others inadequacies in earlier planning cause evaluation results to be of little use in determining whether or not the project has been a success. A post-project assessment would reveal unsuitable use of time, money and techniques, but is often only undertaken where legally required. If not completed, lessons will not be learned and river rehabilitation as a science will not move forward (Holmes 1993; Kondolf 1995; Kondolf & Micheli 1995).

There are a number of reasons why post-project assessment may not be done. First, aquatic ecosystems are complex and vary from year to year, and so positive responses to rehabilitation may be difficult to pinpoint. Second, aquatic ecosystems are slow to respond to change, and so post-project assessment can be slow and expensive (Rutherfurd *et al.* 2000). Third, assessment criteria and techniques are often not considered until after the project is designed and implemented, by which time baseline data of comparable quality, needed to evaluate project success or failure, are rarely available and funds are inadequate. Fourth, sponsoring agencies usually prefer to fund tangible construction projects rather than intangible monitoring and evaluation studies, and so there may be a lack of detailed baseline data for use in a post-project assessment. Fifth, post-project assessment is sometimes abandoned so that any remaining time and money can be spent on implementation. Sixth, general assessment guidelines for rehabilitation projects are not well developed, and so there is little guidance on how to proceed (Kondolf 1995a; Kondolf & Micheli 1995; Foggs & Wells 1998).

Some feel that evaluation procedures should address both biological features and geomorphological (physical) characteristics of the system. To date, most cases of evaluation have focused only on biological criteria. Some specialists in this field recommended that hydrological and geomorphological principles be applied to a much greater extent than in the past to both planning and implementation of rehabilitation (Fogg & Wells 1998; Rutherfurd *et al.* 2000).

Criteria for successful assessment of rehabilitation projects include the following.

- Long-term commitment, in order to truly evaluate project success. Most published data are concerned only with short-term effects of projects (Kondolf 1995a; Friberg *et al.* 1998).
- Measurements need not be made every year, but over a long period in flexible approach. Different types of assessment require different monitoring intervals (Kondolf 1995a).
- There must be a well-defined time limit for completing the assessment. This should take into account the time required for the system to recover.
- Assess project success or failure, *and* what caused these (Rutherfurd *et al.* 2000).
- Acknowledge failure or success and report it. Rehabilitation experiences should be shared in order to increase the limited knowledge base (Kondolf 1995a; Fogg & Wells 1998).
- Document and report the whole assessment process (Fogg & Wells 1998).
- Plan and budget for assessment early in the rehabilitation process (Ladson *et al.* 1999).
- Analytical techniques should complement the monitoring techniques being used.
- Decide whether to assess project completion (outputs) or the influence of the project on the condition of the river (outcomes) (Rutherfurd *et al.* 2000).
- Work out a detailed assessment plan (Ladson *et al.* 1999).
- Choose an assessment process that will convince people about the results of the project (Rutherfurd *et al.* 2000).

- Public participation should be part of the assessment (Brookes 1990).

Guidelines are needed to facilitate the systematic study of past rehabilitation success and failure in order to improve the science (Rutherfurd *et al.* 2000). Objective, scientific assessment is crucial in this last step of the planning and design procedure.

6.5 Summary

A more holistic approach to river rehabilitation is evolving. This should include specialist inputs from government agencies, local municipalities, river rehabilitation agencies, financial institutions, civil and hydraulic engineers, freshwater ecologists, palaeo-ecologists, historical geographers, fluvial geomorphologists, water and navigational agencies, operation companies (e.g. dredging and wood cutting), the general public, riparian landowners, and educational and research institutes. Adopting a holistic approach to rehabilitation adheres to the concept of Integrated Catchment Management, whereby the whole catchment is considered and not just the reach to be rehabilitated.

Successful rehabilitation projects should be based on detailed inter-disciplinary planning and design procedures and consist of nine main steps. These are: developing goals and measurable objectives; river condition analysis; prioritization and project constraints; strategies; study design, techniques and rehabilitation alternatives; feasibility studies; implementation; monitoring and maintenance; and post-project evaluation.

The main barriers to successful rehabilitation include: lack of clear goals or conflict between goals; inappropriate techniques, maintenance or design; lack of established guiding principles; no community involvement; absence of a detailed planning procedure; under funding; planning at reach scale instead of catchment scale; no long-term commitment to ensuring success; lack of post-project evaluation; lack of detailed baseline data; poor understanding of the natural system; addressing the wrong problems; avoidance of responsibilities; and not making project results accessible to the public.

Appropriate goals and objectives are the key to rehabilitation success, while post-project evaluation, up-to-date information, historical analysis, sufficient scientific understanding, participative and repeatable approaches and developing guiding principles are factors that are central to the enhancement of the science of river rehabilitation. In conclusion, failure at any stage in the procedure is likely to cause less than optimal outcomes for the whole project and impede the progress of the science of river rehabilitation.

7 ASSESSING RIVER CONDITION

7.1 Introduction

Assessing river condition is an essential early step in rehabilitation projects (Section 6.4.2), as it places the river into context in terms of its present health. In the past, river condition has been measured mostly by analysing the chemical and physico-chemical characteristics of the water. More recently, a greater range of variables has been recognised as affecting river health (Davis & Finlayson 1999). For instance, a stream that meets water chemistry standards may still be in a degraded state through loss of its riparian vegetation (Ladson & White 2000) or through an unstable bed and banks. A range of biophysical variables, operating at a range of spatial and temporal scales, is now commonly included in assessments of river condition.

In this Chapter, the following topics are addressed: the variables used to measure river condition (Section 7.2), the limitations of assessing a single variable (Section 7.3), and methods for assessing river condition (Section 7.4).

7.2 The variables used to measure river condition

Descriptions of river condition may use biological, hydrological, chemical and physical variables, and sometimes ones depicting human disturbance of the system. Aspects commonly addressed are a range of faunal and floral groups, the flow regime, water chemistry, channel condition and physical habitat, as well as domestic, agricultural, industrial and recreational uses (Coleman & Pettigrove 1999). A more detailed list is provided by Ladson & White (2000):

- hydrology (flow, velocity, flood frequency and magnitude);
- water quality (amount of nutrients, turbidity and pollutants);
- physical habitat (in-stream cover for fish, substratum composition);
- riparian quality (extent of riparian zone, type of species);
- aquatic biology (type and abundance of macro-invertebrates and fish);
- physical form (erosion and sedimentation);
- aesthetics (appearance and recreational appeal).

Building on this list, Rutherfurd *et al*. (2000) suggested that a good rehabilitation project would use the following information:

- the magnitude and duration of different parts of the flow regime;
- water quality;
- structural stability and complexity, including channel sediment movement and the presence of large organic matter such as woody debris;
- the diversity and populations of animals and plants, and the structure of the whole aquatic community;
- the diversity and structure of the riparian community;
- the presence of artificial barriers and their hindrance to the flow of water, sediment and organisms upstream and downstream of the rehabilitation site;
- the presence of lateral structures across the floodplain which form obstructions (levees, channelisation, blocked flood channels).

These variables can be used to measure the present condition of the river, to compare it to some past condition (in order to assess a rehabilitation trajectory back toward natural), or to describe some future condition that may or may not represent reversion toward natural. The majority of rehabilitation projects attempt to revert a river back toward some more natural condition. These require a description, or template, of what that natural condition was, in order to develop goals and detailed objectives.

7.2.1 Building a template of natural condition

Pristine refers to the "primitive" or "original" or "natural" state (Gippel 1999). The pristine state is considered to have existed prior to intensive and widespread disturbance by human cultures. This pre-human disturbance state is an arbitrary reference point and in many cases the extent to which this can be determined relies on the earliest date at which historical data were collected (Gippel 1999). For this reason it is important to state the historical time period to which "pre-disturbance" refers. Historical research from sources such as explorers' diaries, surveyors' notes, archival records, surveyed cross-sections for railway and road bridges, and local interviews can all be used to place the river system and its catchment into a historical context (Davis & Finlayson 1999).

With the historical information collated, information from other sources can be added. Rutherfurd *et al*. (2000) list sources of information other than historical records that can be used for developing the template:

- remnant features left in a field (for example, a secondary channel which has been filled in);
- comparable nearby reaches that are undisturbed and in good condition; these provide an idea of the kinds of habitats and species that once occurred in the project reach;
- known empirical relationships (e.g. channel width v. catchment area);
- generic models of a "good stream" (high biodiversity and diversity of habitat);
- established criteria (water quality, expected assemblages of invertebrates).

The present condition of the river can be compared to the template, in order to assess the degree of degradation. The template can also be used to develop understanding of the causal factors driving this degradation (Davis & Finlayson 1999), and as a reference against which the objectives of a rehabilitation project may be measured.

7.3 Kinds of approaches to assessing river condition

In the past there have been two main approaches to assessing river condition: biological assessments, which use biological indicator species; and channel classification systems, which use descriptions of the physical component of river systems. Neither in isolation provides a comprehensive picture of river condition.

Channel classification systems place a river into a spatial and temporal context within the catchment (Kondolf 1995b), and focus on the dominant geomorphological processes controlling the physical structure of the site in question (Fogg & Wells 1998). Examples are provided by Schuum (1997), Montgomery & Buffington (1993) and Rosgen (1996). These kinds of classification systems do not take into account the dynamic nature of systems, and thus do not provide a complete understanding of the channel and the influences shaping it (Kondolf 1995b). Such classifications are based on physical variables, such as relating sediment transport to channel pattern and stability (Fogg & Wells 1998) and other morphological measurements, such as reach entrenchment, width:depth ratio, sinuosity, the number of channels, and particle size of the slope and bed material (Rosgen (1996 cited by Fogg & Wells 1998). Attempts are made to describe variability in major river forms, and processes across catchments as determined by channel geometry (size and shape) and the presence of associated landforms (Brierly 1999).

Biological indicators are used to infer the ecological health of a river while also providing an indirect measure of water quality (Rutherfurd *et al*. 2000). Values are attached to whether or not specific floral or faunal taxa are present or absent. Data are collected using standardised methods, and so information between sites and samples can be compared. The methods assume that diversity is important (five species present is better than one) (Rutherfurd *et al*. 2000), and rank sensitive animals characteristic of undisturbed sites higher than hardy species characteristic of disturbed sites. Examples of these methods include AusrivAS (Simpson *et al*. 1997), SASS (South African Scoring System) (Dickens & Graham 2001), SIGNAL (Stream Invertebrate Grade Number – Average Level) (Chessman 1995), ORCM (Oak Ridge Chinook Salmon Model) (FERC 1996) and SALMOD (Salmonid Population Model) (Bartholow *et al*. 1997).

The reliance on a single indicator limits the ability to assess river condition with confidence, since different indicators will respond to different environmental stresses in different ways (Coleman & Pettigrove 1999). A better appreciation of river condition can be gained using a suite of indicators.

7.4 Methods for assessing river condition

Methods for measuring the condition of rivers have been developed for two main reasons. First, they can be used to provide a snapshot of the condition of the stream. Such methods may consist of once-off measurements that compare the present condition to a chosen reference (usually natural) condition. Second, they can be used to provide long term data sets that will aid in the setting of objectives, planning and reporting for on-going adaptive management of rivers and their catchments (Ladson & White 2000). Most of the methods were developed to facilitate integrated management of river systems. They can be used to measure river condition before and after intervention, or to compare expected with observed changes in condition in order to quantify the environmental consequences of management decisions (Ladson & White 2000). No method presently available addresses all aspects of river condition and, for a comprehensive assessment; it may be most useful to employ a range.

Five approaches to assessing river condition are outlined here. All measure one or more variables, differing in their choice of these and their methods for quantifying them. Some are holistic approaches, while others focus on a single aspect of degradation. The five methods are from Australia and the United States: the Rivercare Approach, the State of the Rivers Method, the Index of Stream Condition, the National Water Quality Inventory and the State of the Environment Report. The two most comprehensive in terms of their coverage of the different categories of variables are the Index of Stream Condition and the National Water Quality Inventory.

7.4.1 The Rivercare Approach

This approach was developed by the New South Wales Department of Land and Water Conservation in Australia, and is based on an objective of rehabilitating a river to becoming well vegetated with a stable channel. The method is summarised by Raine & Gardner (1995 cited by Rutherfurd *et al*. 2000). Condition is tagged on aerial photographs at the reach scale along the rivers of concern. The condition of the riparian vegetation and the stability of the banks are ranked as bad (red) for a poorly vegetated, unstable channel, average (yellow) or good (green) for a well-vegetated and stable channel. This traffic light approach (Rutherfurd *et al* 2000) can be applied across entire catchments to help prioritise reaches in need of intervention. The Rivercare approach was designed for use in the northern coastal systems of New South Wales, and is specific to these rivers. It does not directly consider water quality or flow regulation, but rather provides a tool for communities wishing to manage stream erosion and deposition in short reaches.

7.4.2 The State of the Rivers Method

The State of the Rivers Method was developed by Anderson (1993, cited by Rutherfurd 2000) for use by the Queensland government, Australia. The method assumes that physical condition is an acceptable surrogate for ecological condition, and assesses this for complete river systems. Present condition is compared to a presumed natural condition using local undisturbed sites as a reference (Rutherfurd *et al*. 2000). Descriptive data are manually recorded onto prepared data sheets and than entered into a GIS program for analysis. Each site is assessed for the status of the channel and aquatic habitat, bank condition, bed and bar condition, and riparian and aquatic vegetation. The method relies on how accurately the persons collecting the data can interpret what they see and how well the algorithms in the computer programs define the natural condition of each reach. The State of the Rivers method provides a structured approach to recording information about river sites, and has a large and easy-to-use computerised analysis program, which makes prioritisation and planning of reaches easy for rehabilitation projects (Rutherfurd *et al*. 2000). The method does not consider the issues of water quality or regulation, or interaction between reaches and does not assess the state of the aquatic biota (Ladson & White 2000). Since it assesses individual sites rather than the interaction between sites, it cannot provide reasons for the condition of the reaches it assesses (Rutherfurd *et al*. 2000).

7.4.3 The Index of Stream Condition

The Index of Stream Condition was developed for use in Victoria, Australia to assess the condition of the rivers throughout that State. It is used for setting river management goals (Ladson & White 2000), and for monitoring the effectiveness of management interventions over a long period. It may not easily be transferable to rivers outside Victoria (Rutherfurd *et al*. 2000). The Index is based on the following five components and sub-components of river condition (Ladson & White 2000).

- Hydrology. Sub-components: change in volume and seasonality of flow from the expected natural condition.
- Physical form. Sub-components: bank stability, bed erosion or aggradation, the presence of artificial and natural barriers.
- Riparian vegetation. Sub-components: the type of species, spatial extent, width and condition, presence of wetlands and ox-bow lakes.

- Water quality. Sub-components: total phosphorus, turbidity, conductivity and pH.
- Aquatic life. Sub-component: the number and type of aquatic invertebrates based on the SIGNAL index of Chessman (1995).

The Index is compiled by initially giving each sub-component a rating of one to ten based on its condition compared to natural. It is similar to the State of the Rivers method, in that it employs a subjective rating system, but unlike it in that it also includes some easy-to-measure physical characteristics (Rutherfurd *et al*. 2000). Sub-components, and then components are totalled to present a summary of overall condition, with the individual assessment scores retained so that the user can identify which component is contributing most to a poor score. The approach is useful for highlighting problems but cannot provide detailed information on the nature or cause of a highlighted problem (Ladson & White 2000), or the direction of change (Rutherfurd *et al*. 2000). Due to its lack of detail, the approach is suitable for monitoring purposes but not for scientific hypothesis testing (Rutherfurd *et al*. 2000).

7.4.4 The National Water Quality Inventory

The following information is from Ladson & White (2000). The National Water Quality Inventory was a response to the U.S. Clean Water Act in 1972. Its purpose is to aid protection of the physical, chemical and biological integrity of rivers. The measurement of river condition varies between the participating States, since they select their own standards depending upon the intended beneficial users of the particular river system. This may be the "protection and propagation of fish, shellfish and wildlife" or to "provide for recreation in and on the water". A suite of criteria, for instance, dissolved oxygen, and the presence or absence of certain indicator species, is assessed regularly. The method thus has the potential to track management performance, by revealing any improvement or loss of river condition in the long term. The main limitation of the method is the lack of consistency between participating jurisdictions, which makes comparison between sites and States difficult.

7.4.5 State of the Environment Reporting

The State of the Environment approach was developed in Victoria, Australia, to provide an Index of Aquatic Environmental Quality (IAEQ). It includes an assessment of:

- physical water quality (turbidity, suspended solids, conductivity);
- chemical water quality (biochemical oxygen demand, pH, phosphorus, nitrogen and toxicants);
- amount of toxicants in sediments (metals and hydrocarbons);
- macro-invertebrates (abundance and diversity of expected and observed species assemblages);
- fish (type, abundance and diversity, compared to expected natural levels);
- extent and quality of the riparian vegetation.

Each category is given a score on a five-point scale according to a list of provided descriptions. The resulting report represents a one-off assessment based on the difference between the present and natural condition of selected rivers (Ladson & White 2000). Analysis relies heavily on historical data, which are not available for many rivers.

7.5 Summary

Assessing river condition is an essential step in rehabilitation projects. A good project will collect condition information on a range of chemical, physical, physico-chemical and biological variables. No existing method can fully describe river condition. Methods differ in their focus and the ways data are collected. The kind of condition assessments that are most useful in rehabilitation projects are those that compare the existing measured condition of a river to its natural condition, or to its prior state if that is less than natural, in order to measure how degraded or close to natural the system is. This prior, or reference, condition, can be used to guide rehabilitation activities.

8. IMPLEMENTATION: METHODS AND MATERIALS

8.1 Introduction

Conventional river engineering practices, sometime called “hard” engineering, have brought many benefits, but also resulted in significant social and environmental impacts and new kinds of management problems. River channels have been straightened, raised, lowered, lined with concrete, narrowed, widened, diverted and dammed in order to achieve objectives such as flood control, navigation or bank stabilisation, with little or no regard for the social or ecological implications of changing the ecosystems. The degradation of the rivers has led to increasing public concern, and a growing number of attempts to restore or rehabilitate them (Rosgen 1994; Kondolf 1996). Such activities can reduce adverse environmental impacts while restoring ecological, aesthetic and recreational values, through the development of innovative approaches, or modifications to existing ones. These may involve bio-engineering techniques that integrate biological and engineering aspects, resulting in a ‘softer’ more natural appearance for rehabilitated rivers. Some of the more sensitive bio-engineering techniques, for instance, include the development of two-stage channels; off-channel wetland filters; flood-relief channels; new pools and riffles; and instream habitat devices. Modifications more allied to ‘hard’ engineering structures include gabion baskets or riprap to support bank vegetation and soils (Fogg & Wells 1998; Gore 1996; Kondolf 1996; Luger 1998; Hoitsma 1999; Rutherfurd *et al*. 2000).

In this Chapter, a variety of tools and techniques for river rehabilitation are listed, for environmentally-sensitive river maintenance (Section 8.2), for soft bio-engineering approaches (Section 8.3), and for mitigation of hard engineering practices (Section 8.4). The option for allowing natural recovery is also briefly addressed (Section 8.5). The tools and techniques are reported but not evaluated. Different ones will apply in different situations.

8.2 Environmentally sensitive river maintenance activities

8.2.1 Reduce the frequency of maintenance practices

Regular river maintenance does not allow a degraded river to adjust to a new equilibrium, whether this be recovery back toward natural or change to some new state (Brookes 1997; Luger 1998). Such regular maintenance may result in continuing bank instability, loss of habitat, unstable substrata and encroachment of alien vegetation. Reduced maintenance, sometimes called the ‘do minimum’ option, involves reducing the frequency of maintenance actions so that a new equilibrium can gradually develop. Maintenance costs are reduced, but it is difficult to judge the frequency of intervention that will allow the river to achieve an acceptable equilibrium condition while still addressing other public needs.

8.2.2 Modify one bank

According to Holmes & Newbold (1989) the earliest guidelines for sensitive modification of channels recommended leaving one bank untouched. It was suggested that any channel enlargement be done on one side only, and the existing channel alignment followed. The untouched bank could then provide a source of plant and animal species for recolonisation of the modified bank. Hydraulic considerations may dictate which bank should be modified: it may be better to modify the insides of bends and conserve vegetation on the outsides for bank stability (Shields 1982a, b, cited in Luger 1998).

8.2.3 Reduce or eliminate dredging operations

Channels may be dredged to deepen or widen them, or to remove accumulated sediments (Brookes 1997; Rutherfurd *et al*. 2000). A large range of machinery is available for this including:

- hydraulic draglines attached to land-based vehicles, which extract sand with a large bucket that is dragged across the riverbed;
- hydraulic suction dredgers that suck sand from the riverbed;
- bulldozers that push sediments from the riverbed.

Dredging can cause upstream and downstream bed degradation, increase or decrease water depth, or damage the riparian zone through the impact of the machinery employed. Bed degradation is normally in the form of bed lowering, generally equal to the depth of the sediment removed from it. Bed lowering can also lead to destabilization of the banks, lowering of the water table and coarsening of the bed. The latter may result from the development of an armour layer of sediments larger in size than elsewhere in the channel (Rutherfurd *et al*. 2000). Dredging only part of the channel reduces the damage, thus ensuring some habitats and associated species remain. Hydraulic characteristics such as unsteady and non-uniform flow that lead to erosion on inside bends, should be kept in mind in determining where partial dredging should be directed, thereby ensuring stability across a wide range of flows (Brookes 1997). The rate of extraction has to be slow enough to allow the river to stabilize. Rapid deepening of the channel could lead to increased bank erosion (slumping) and thus channel destabilization, whereas a slower rate of extraction would give the channel a better chance of recovery (Rutherfurd *et al*. 2000).

8.2.4 Careful disposal of spoil

Spoil produced when channels are dredged should be disposed of with the following in mind (Luger 1998).

- Riverbanks and riparian zones often contain diverse communities of species that do not occur on adjacent drier land. Spoil should not be dumped in these areas, because these communities play a vital role in the health of the whole river ecosystem. Additionally, spoil dumped in such areas could wash or leach back into the river.
- Large riparian trees should not have spoil spread around them.
- Hollows and other irregularities in the surface of adjacent floodplains should not be filled. These hollows impart physical heterogeneity to the landscape, allowing a greater diversity of plant and animal species to exist.
- Ditch systems and old ponds should not be filled with spoil because they may constitute rare habitats and support combinations of wetland and aquatic fauna and flora.

8.2.5 Control stock access

Good riparian-grazing practices can effectively reduce bank erosion, and improve water quality and aquatic habitats (Karssies & Prosser 1999; Rutherfurd *et al* 2000). By controlling access of livestock to the river, the development of stock tracks (direct pathways for runoff from the paddock into the river) and bank trampling can be avoided. The most direct and effective control mechanism is fencing. All or most of the riverbank can be fenced off, leaving a filter strip along the river to trap sediments and nutrients draining toward it. Restricted access points could remain, or alternative off-stream watering points supplied. Such off stream points could use a range of pumping devices, such as electric pumps or wind pumps, to suit the environment and the landholder's needs.

8.2.6 Reduce the frequency and intensity of mowing

Mowing is the main method of managing vegetation along riverbanks and adjacent land (Coux & Welcomme 1998; Luger 1998). Good practice approaches include the following.

- Cut one bank per year. This will enhance floral and faunal diversity and be financially beneficial whilst preventing domination by invasive plants.
- Always leave a strip of vegetation at the bottom of the bank. This provides vital habitat, cover and food at the water's edge.
- Leave narrow strips of vegetation uncut up the bank, even on the mowed side. This provides remnants of habitat, and passage for small animals to the water until re-growth occurs.
- Leave isolated seedling trees and do not mow wider parts unless flood capacity will be compromised.
- Leave patches of rare or important species. A site-specific management plan should be provided for these.
- Always leave at least 50% of shrubs or mature trees unmowed or uncut in any one year.

The key to successful riparian-zone management is to retain as much native vegetation as possible and to leave particularly sensitive areas intact.

8.2.7 Maintain or establish riparian and aquatic vegetation

Riparian and instream vegetation is often cut back or controlled in other ways because other functional uses of a river are compromised when its growth is excessive. From an engineering point of view, aquatic vegetation can clog navigable channels and increase channel roughness, thereby resulting in a loss of capacity to carry high flows. However, the total removal of aquatic and riparian vegetation can lead to other problems, and a more acceptable approach might be to manage for a reasonable amount of water and bankside plants.

Maintaining or establishing vegetation in, or on the banks of, watercourses considerably enhances river landscapes (Coux & Welcomme 1998). According to Holmes & Newbold (1989) and Luger (1998), riparian and aquatic vegetation perform the following valuable roles in aquatic ecosystems.

- Tree roots increase the tensional strength of the river bank and protect it against erosion.
- Riparian vegetation thus stabilizes floodplains and channels and reduces erosion.
- Shade cast from riverside trees controls aquatic vegetation as well as enhancing habitat value.
- Trees in the floodplain increase the roughness factor, and thus resistance to flow, during floods. The reduction in velocity decreases shear stress on the channel bank, which leads to the deposition of fine sediments and elevation of water levels at a given discharge. Vegetation may thus attenuate flood flows by slowing velocities and causing water to spread higher on the floodplain (Friedman & Auable 2000).

- Healthy stands of native riparian vegetation can prevent invasive species becoming established, and nutrient levels in the stream reaching levels that promote algal blooms blooms.
- Aquatic and riparian vegetation provides food and habitats for a range of organisms.
- Trees provide microclimates for organisms that require a damp shaded environment.
- Aquatic plants oxygenate and purify the water, and supply breeding, nesting, spawning and refuge areas for many species.
- Well-anchored, submerged aquatic vegetation stabilises channel beds.

The more varied the structure and compositions of the plant communities the greater the diversity of other organisms.

8.2.8 Control aquatic weeds

Waterweeds may grow prolifically, especially in nutrient-rich warm waters, and need to be cut to free channels and create more diverse habitats (Holmes & Newbold 1989; Coux & Welcomme 1998; Luger 1998). They may be cut with scythes or machines (Brookes 1997) or sprayed with herbicides. For some species, biological control measures are also available. Trees can be planted to produce shade that controls weeds. Cut weeds should be removed from the river environ, to avoid blockage of downstream structures and eutrophication and deoxygenation of the water as they decay.

8.2.9 Create sediment traps

Construction works often introduce fine sediments into channels. These raise the level of the riverbed, smothering habitats and food needed by aquatic and riparian species (Kern 1992b). Temporary sand traps can be used to reduce these inputs during rehabilitation work. These retain the fines, reducing downstream deposition and the associated maintenance and ecological costs. Traps should be located where machines can easily reach them to remove the sediments. Sediment inputs can be further reduced by avoiding maintenance work during periods of high flow; limiting bed and bank disruption; minimising the excavated area and containing the length of the rehabilitation activity (Luger 1998).

8.3 Soft bioengineering approaches

8.3.1 Stream bank protection

Bank erosion is a natural process in parts of most rivers. River flows erode banks on concave bends, transport the bank material downstream and deposit it on convex bends. These alternating erosional and depositional processes maintain meander patterns in the channel, perhaps resulting in one landowner losing land whilst the landowner on the opposite bank gains some. Whether due to natural causes or human activities, land losses of this kind may create the demand for bank protection (Henderson 1986; Holmes & Newbold 1989; Kondolf 1996).

There has been extensive research into methods of streambank protection that work with nature rather than against it. Soil bioengineering is the process of combining biological and engineering concepts to control processes such as erosion and sediment movement. Rehabilitation techniques employing this approach usually use a mixture of natural wood and stone, living vegetation and man-made materials (Brookes & Hanbury 1990; Cairns 1995; Gore *et al*. 1995; Fogg & Wells 1998; Luger 1998; Hoitsma 1999; Shaw 1999). These have the following advantages.

- Food and habitats are provided for native flora and fauna (Henderson 1986).
- They are aesthetically superior to slabs of concrete.
- Substantial cost savings: the use of vegetation and other nonstructural materials can reduce the costs of construction or off-site mitigation. However, maintenance and repair costs need to be considered (Henderson 1986; Hoitsma 1999).
- Can be used to conceal or protect water-retention devices (Brookes & Hanbury 1990).
- Structures involving living vegetation tend to be self-healing when damaged.
- Natural deterioration of the materials will not degrade the ecosystem.
- Bioengineering measures are labour intensive, but their sociological and economic benefits are beginning to make these measures more desirable as alternatives for bank stabilization (Luger 1998).
- The structures provide immediate bank stability, when properly installed.
- They protect new vegetation until it is established and reduce soil slumping as the vegetation removes excess moisture from the banks.

Bioengineered bank protection systems need to take into account the complex interactions between hydrological, hydraulic, geomorphological, geotechnical, vegetative and construction factors. According to Henderson (1986), bioengineering systems involve placing materials such as geotextile fabric or mesh, terraced geocells, interlocking concrete blocks, fabric-encapsulated soil (FES), stone, rock, wood or vegetation in contact with the bank.

Geotextile fabrics are woven netting made of synthetic (e.g. polymer matting, nylon) or natural fibre (e.g. coconut fibres or coir cotton, jute and sisal). Synthetic fabrics are not favoured because they are not biodegradable and may pose a threat to organisms if they are washed out. Coir fabrics are popular because they have high tensile strengths, do not rot easily and can withstand high current speeds. Jute netting apparently functions less well in high flows (Riley 1998). Geotextiles can be temporary (biodegradable fabric or mesh), functioning as short-term bank protection until dense vegetative cover has developed, or permanent (stable fabric or mesh), functioning effectively as a long-term bank protection (Luger 1998; Riley 1998). Once shaded from sunlight by developing vegetation, the stable fabric usually remains *in situ* to protect the bank should the vegetation be damaged. The lifetime of biodegradable geotextile fabric layers is estimated at 5-7 years, although Hoitsma (1999) feels 2-4 years may be more realistic. Rapid establishment of a diverse plant community can be promoted by using a loose-weave geotextile, planting vegetation in pockets in it, and replacing the original topsoil. However, the relationship between decaying fabric and plant establishment is site specific.

Geocells are cellular confinement membranes, stapled together in terraced rows. They are made of synthetic materials filled with a mixture of compacted soil and cobble. The cells are installed within the banks. Seeds are planted on the exposed surfaces of the geocells and are normally covered with a geotextile fabric such as coir fabric. The web of soil-filled synthetic cells, the interlocking plant roots and the aboveground vegetation impart a high resistance to erosive flows. The use of synthetic materials results in relatively expensive structures (Hoitsma 1999).

A variety of interlocking concrete block systems are available, such as “Grinaker Waterloffel”, “Terraforce”, “Terrafix”, “Loffelstein” and “Armorflex”. These blocks can contain soil and where possible are planted with vegetation for a more attractive environment. These cellular blocks are mostly used to stabilize and protect evenly graded slopes, usually on channel bends. They reduce current speeds and enhance oxygen levels through friction of the water against the wall, are aesthetically softer than concrete, and allow growth of some vegetative cover (Luger 1998). The Grinaker Waterloffel incorporates a compartment wall and two wings that lock the individual units together, thereby creating a solid concrete face that protects the underlying bank from scouring and erosion. Their closed design, however, restricts the roots to a shallow (less than 250 mm) pot and the lack of drainage holes prevents the drainage of water (Luger 1998). Successful use of these interlocking systems usually requires good foundations, such as bedrock or gabions, as well as an adequate drainage system to avoid any imbalance of pressure behind and in front of the walls (Precast 1992). Successful plant cover can only be achieved if the depth of the soil both within and below the blocks is adequate. These interlocking systems are frequently laid on a filter layer (e.g. a geotextile fabric, or graded layers of granular material) to prevent soil loss, but this may restrict root penetration. An alternative method uses "Grasscrete", "Dymex", "Gobimat" or "Hyson" cells. These cells are formed by pouring concrete *in situ* into plastic formers, resulting in a high surface area of concrete for a given volume. The plastic formers can be filled with topsoil and planted with vegetation. A disadvantage is that the smoothness of the concrete reduces friction, which may increase scour of soil and thus inhibit plant growth. These systems offer few environmental benefits other than being preferable to solid concrete (Luger 1998).

The upper slope or midsection of the stream bank can be strengthened and protected from erosion by fabric-encapsulated soil (FES) lifts or geogrids. These can be defined as coarsely textured gravel-like sediments or soils wrapped around by or encapsulated within two layers of biodegradable geotextile coir fabric. The geogrids are between 0.9 m and 2.3 m wide and 0.3 m to 0.7 m high, and are placed perpendicular to the bank on a slope of between 1:1 and 3:1. The inner layer, a non-woven mat of coconut fibers held together with a polypropylene thread mesh, prevents fine sediment from piping through the coarser outer layer. The outer layer, a heavy weight coir fabric of twisted coconut fibers woven into a strong mesh, provides structural integrity to the lift and therefore the bank. Lift dimensions (e.g. embankment length, lift height and fabric overlap distance) are determined through slope stability analysis. The geogrids can be employed in conjunction with a rock toe foundation overlain by a gravel filter drain system, in order to withstand scouring and undercutting at bends. Long-term stabilization of the geogrids is ensured by planting deep-rooting cuttings (e.g. willow, dogwood) between them, and placing grass and forb seeds through slits beneath the coir fabric layers on the face and top of each of them. The geogrids will be well colonized by riparian plants by the time the fabric disintegrates. The geogrids are designed to withstand bank shear stresses prior to the establishment of vegetation. Fiber-schines - cylindrical fiber bundles staked to a bank with cuttings that are inserted through or into the material - are also used in stream bank treatments (Miller 1996b; Fogg & Wells 1998; Hoitsma 1999).

Erosion of the bank toe or degradation of the streambed often causes failure of a steep bank. Stabilization of the bank toe can stop channel migration, bed degradation and bank undermining.

According to Luger (1998) and Hoitsma (1999) rock, stone or log foundations may be used to protect the bank toe and to provide a stable platform for the kinds of bioengineering measures mentioned above. Rock or stone foundations are preferred materials, and help to prevent bank undermining by deflecting flow from the bank, promoting the deposition of sediment on the bank behind the toe, and preserving the vegetation above it. Toe rock should be inserted into solid ground, and be of a weight relevant to local bank pressure points associated with bank failure (Henderson 1986; Miller 1996b; Shaw 1999).

Bank instability can also be addressed by stabilizing the entire bank with rock. Protection of the toe is crucial, with rocks staggered and placed to maximize interlocking between the rocks. The resulting variations in rock height impart a natural appearance. To ensure long-term structural integrity, the voids in and around the rocks can be filled with smaller materials. Keys to success for this kind of stabilization work are: varying the bank slopes; using a range of rock sizes; varying the height of the rock work up the bank; placement of a toe rock foundation; ensuring water flows over rather than under the finished product; and utilizing any existing natural rock found at the site (Shaw 1999).

Soil bioengineering systems usually employ plant materials in their designs. The long-term stability of bioengineering treatments depends on establishing a dense plant community that binds soils with root systems and provides protection against erosive forces through exposed shoots, stems and leaves (Miller 1996b). It is important to use appropriate plant species, which is most easily done by choosing riparian species native to the rehabilitation site. As the distribution of plant species up the bank is related to the frequency with which they are inundated, the depth to the water table and soil-moisture levels, understanding the hydrological and hydraulic features of the site is vital (Cairns 1995; Rutherfurd *et al.* 2000).

Live woody cuttings, roof plugs, containerized stock, tublings and cuttings, bare-root plants and poles of readily sprouting species are commonly used in rehabilitation (Gray & Leiser 1982; Fogg & Wells 1998; Riley 1998). Tublings are plants grown in single paper or plastic tubes (Gray & Leiser 1982; Miller 1996b). Bare-root seedlings are normally packed in bundles with their roots wrapped in a moist absorbent material and plastic or wrapping paper to keep the moisture in. Tube plants, bare root plants, cuttings and seeds are favoured over container stock for a number of reasons in addition to expense. Container stock grows more slowly than other plant stocks because growth of the roots may temporarily be confined to the nursery soil from the pot. They also take a long time to acclimate to their new surroundings, whereas cuttings or seeds are forced to immediately adapt to their new environment (Riley 1998). Woody plants may be established either by embedding individual cuttings or containerized plants in tubling containers, tree-tubes or small pots into eroding banks. In addition, there are several techniques available that employ a large number of cuttings arranged or planted in bundles (called fascines or wattles) or layers (brush layering), which can be secured to riverbanks. A contour fascine or wattle, defined as a cigar-shaped bundle of tied cuttings (e.g. willows), can be planted and partially buried in shallow trenches. Fascines are laid end to end on the contour of a slope and are buried only a few centimeters below the grade of the slope. In the case of brush layering, cuttings are layered on terraces perpendicular to the slope so that they will root into the bank or hillside. Only the growing tips protrude from the slope because the live brush is covered with soil. Brush layering is used in combination with pole cuttings in the active channel (Gray & Leiser 1982; Fogg & Wells 1998; Riley 1998).

Another well-established technique using live plants is the use of brush matting, brush mattresses or woven mats in stream bank protection and erosion control. It is employed in conjunction with other structural measures (e.g. riprap or groynes) to create areas that can temporarily resist inundation but not scouring and undercutting (Gray & Leiser 1982; Fogg & Wells 1998). Brush matting differs from fascines and brush layering in that it is essentially a mulch of hardwood cuttings (e.g. willow, elm, maple) laid on the face of a bank, and interwoven or fastened down with jute wire or cord held in place by stakes. The brush is laid flat on the bank with the ends pointed upstream, usually toward the middle of the bank. It can extend to the top of the bank, however, for floodplain slopes of 15^{o} or less (Gray & Leiser 1982). Unrooted, heavy cuttings are inserted or planted before or after matting installation, though difficulties such as planting through the brush matting and seasonal requirements for planting need to be considered (Gray & Leiser 1982). The technique provides direct protection from erosion, facilitates trapping of sediments, and allows roots to grow, thus strengthening the soil and providing direct protection from erosive flows. Vegetative growth above the ground reduces over-bank water velocities, further protecting the bank from high flows. Banks should be sloped according to the terrain and soil texture. As a general guideline, slopes should be 2:1 (e.g. 2 m horizontal to 1 m vertical) or flatter, because establishing plants on steeper banks is difficult (Bowie 1982; Miller 1996b).

The use of vegetation and nonstructural materials can reduce the cost of rehabilitation, although there are still costs involved in properly sustaining, improving and extending the lives of the plants. This kind of bank protection is most effective when used in an appropriate combination with more traditional structural materials (Gore *et al*. 1995).

Soil bioengineering systems are not suitable for every rehabilitation project. By demonstrating the integrity of these treatments, developing design and implementation guidelines and continually improving their designs, however, their popularity as rehabilitation alternatives is growing (Fogg & Wells 1998; Hoitsma 1999).

8.3.2 Re-vegetation

Rehabilitation of banks may be attempted with vegetation alone, with no use of structural aids. This is most successful if based on the natural patterns of distribution of plant species in the aquatic and riparian areas. Vegetation zones linked to inundation levels can be detected from any nearby natural plant cover. Large trees with deep root systems occur at the top of the active channel bank and into the macro-channel (flood channel) while shrubs, herbaceous perennials and grasses, reeds and sedges, and mosses and ferns occur in distinct zones closer to the water (Boucher 2002). Each zone consists of an assemblage of species adapted to survive the particular inundation and exposure patterns pertaining to that location. Planting the correct species in each zone is critical to the success of the rehabilitation effort.

Key factors to consider are:

- complete a thorough site analysis of, *inter alia*, microclimate, soils, site conditions and the vegetation immediately adjacent to the site (Gray & Leiser 1982);
- identify the most important erosion processes operating at the project site;
- match plants species to the identified erosion areas (Shaw 1999; Rutherfurd *et al*. 2000), selecting adaptable, tough, competitive and trouble free species where possible;

- adopt an appropriate planting technique and plant in the appropriate season;
- in terms of species establishment, re-vegetation usually begins with pioneer species that easily take root from cuttings or start from seed, and these in turn create the environment for a succession of other species;
- use an irrigation system until the plants are established (Riley 1998);
- ensure sites are reasonably stable for at least one growing season after planting to allow plants to establish themselves, managing the project (Gray & Leiser 1982), and monitoring developments carefully (Gore 1985).

If the appropriate species are already growing at the site, re-vegetation can sometimes be best accomplished by simply allowing these to re-colonise the area. This would achieve good ecological diversity as well as being economical. An example would be fencing off livestock from riparian corridors to allow the recovery of native riparian species (Riley 1998). The use of vegetation for rehabilitation may be inappropriate in areas where current speeds are high and banks steep. Plants on their own are rarely effective in these areas and should be combined with other bank stabilization measures (Coux and Welcomme 1998).

Soil bio-engineering techniques such as the above can have the following disadvantages or problems.

- Those relying on vegetation are particularly vulnerable to washout by floods. Plantings may be destroyed if subjected to high flows before they are fully established. It may take years for vegetation to become well established (Fogg & Wells 1998; Moses & Morris 1998).
- Many native plants used are not riverine or even riparian. They could lack the extensive rooting systems needed for bank protection, fail to provide food to stream organisms through dropping the wrong kinds of leaves into the water in the wrong season, and fail to provide suitable shelter for river and bank organisms.
- Exotic, invasive weeds (e.g. kikuyu grass, willows) are sometimes used because of their ease of establishment, rapid growth rates, fibrous root systems, and ability to survive in floods. Where maintenance is poor, these can rapidly replace indigenous vegetation (Luger & Davies 1993; Hicks *et al*. 1999).
- Environmentally sensitive treatments may require more space so that, for instance, banks can be sloped adequately and vegetation zones established up the banks. Much rehabilitation takes place in urban areas where space is limited (Gore *et al.* 1995; Kondolf 1996).
- Techniques using plants may be inappropriate where the bank materials are non-cohesive, and are most successful in sunny, open environments. Rehabilitation of this kind on cooler and well-shaded banks may be unsuccessful (Luger & Davies 1993; Riley 1998).
- Live plantings are more prone to be damaged by floods than hard engineering structures that are protected by steel or chain mesh structures. Such damage may be viewed as total failure, impeding follow-up repair work and maintenance of the vegetation (Starr 1999).

8.4 Hard engineering practices and their mitigation

Hard engineering approaches for channel stabilization range from simple structures such as boulder riprap to complex structural solutions such as concrete revetments (Gore *et al*. 1995). According to Foggs & Wells (1998), stabilization methods can be grouped into three categories: surface armour, indirect and vegetative. Armour, defined as protective material in direct contact with the streambank, can consist of stone, self-adjusting armour (e.g. sacks, blocks, rubble), rigid armour (e.g. concrete) or flexible mattresses (gabions, concrete blocks). Indirect methods consist of structures that extend into the stream channel and redirect the flow away from the area of concern. Methods that use vegetation, discussed in Section 8.3, can function as either armour, or indirect protection, or a combination of both. Some common approaches to streambank and instream modifications, and possible mitigation of their impacts, are detailed below. Further details of the rehabilitation of banks, riparian zones and floodplains are given in Chapter 9, and of the channel in Chapter 10.

8.4.1 Bank zone

See also Chapter 9.

Structural control

Structural controls of banks usually include rock or boulder riprap, gabion revetments or concrete sacks (Schultze & Wilcox 1985). Riprap, the most commonly used bank protection, consists of a layer of large boulders of different sizes placed on a shaped slope (Gore *et al.* 1995). Its advantage over concrete is that it aesthetically more pleasing, and it provides spaces, crevices and rough surfaces where organisms can shelter, sediments accumulate and plants grow. Thus more primary productivity can occur than on concrete slabs, and there is a greater diversity of substrata and thus habitats and species. Its main disadvantages are high transport and handling costs (Luger 1998).

Gabions are flexible structures that can be used for both bank and bed protection and to construct weirs (Luger 1998). Gabions are normally wire-mesh baskets filled with rocks, and allow use of small stones that would otherwise be unstable in the flow. Basket gabions are used to protect steep vertical banks, and mattress or blanket gabions offer slope protection (Schultze & Wilcox 1985). Silt soon collects in the interstitial spaces between the stones, aquatic organisms colonise these areas and plants grow over the wire mesh. Their main disadvantage is that after about ten years the wire cage corrodes or fails through metal fatigue and the cage breaks open, depositing broken and rusty wire on the banks and in the river (Luger 1998).

Revetments are primarily used on the outer side of channel bends where velocities can be high enough to erode the bank (Maynord 1989). They prevent lateral migration of meandering rivers, and erosion, especially at the toe of the bank (Harvey & Sing 1989). They may be made from concrete, aluminum sheet, pipes, wires or used automobile tyres. With a little ingenuity and the use of some wood, they can provide significant habitat for many benthic organisms and plants (Gore *et al.* 1995).

The above-mentioned structural measures impart higher initial control of erosion than soil bioengineering techniques and appear to have a longer project life, but they may be costlier and many provide little in the way of habitat. Additionally, local flow conditions may be altered by hardening an isolated stretch of bank. Currents may be pushed in new directions, and new areas of erosion initiated (Kondolf 1996).

Protection or retaining walls, or barriers

Protection walls can take the form of berms or banks constructed of earth, boulders, gabions, concrete, or interconnecting blocks. Use of interlocking blocks instead of solid concrete will allow some lateral movement of water and organisms between the river and its banks (Luger & Davies 1993). Such barriers on both banks retain the flow within a predetermined channel and are usually used as a flood mitigation measure. Where barriers are confined to one bank, flood waters can spill over the other one, perhaps on purpose to ensure flooding of a floodplain (Crowther Campbell & Associates 2000).

Lunker and A-jack installations

Lunker and A-jack installations are installed at the bottom of the bank to protect the bank toe. They are normally used with rocks and vegetation to stabilize the water line and the upper bank. A lunker is a large wooden or plastic box (approx 2 m x 1 m) with both ends and the stream side open. Plants (e.g. cuttings) and large rocks are placed into a trench behind the lunker with a geofabric mesh called fibredam. The lunker structure is empty and fibredam reduces soil movement through the structure. Riprap and soil are then placed behind and above the lunkers while the bank is sloped over the lunkers and seeded (Roseboom 1994). The lunker is aesthetically acceptable, cheap to install and increases aquatic habitat. During the first year, establishment of vegetation is critical to the success of the technique and labour costs for construction and installation might be high.

A-jacks are interlocking concrete jacks that are more stable than riprap and so better able to withstand heavy storm conditions. The rows of A-jacks interlock, and dense root systems can grow through the interstices. Large stones may be laid over the A-jack rows for aesthetic purposes. The lunker technique, combined with A-jacks, rocks and upper bank vegetation has proved very durable (Roseboom 1994).

8.4.2 Aquatic zone

See also Chapter 10.

Full-width structures

Grade-control, full-width, or drop structures are low bed-raising structures that extend across the entire width of the channel, although some incorporate a notch to concentrate low flows (Brookes *et al*. 1996; Rutherfurd *et al*. 2000). They control the longitudinal profile of a river through a local dissipation of energy, thus allowing a flatter slope with occasional drops to develop. They are overtopped by water under most flow conditions. They may include check dams, weirs and sills, and increase habitat diversity through the creation of pool and riffle characteristics. They can be built from logs, boulders, gabions, rock, metal sheet piling, or concrete (Brookes *et al*. 1996; Shields *et al*. 1995b, 1997), and have the following attributes:

- a backwater pool is created upstream of the structure and a scour pool immediately down stream, increasing habitat and thus biological diversity;
- bars or riffle-like features develop downstream of the scour pool, again increasing habitat diversity;
- variability of flow is increased (Luger 1998);
- spawning gravels and fine sediments are trapped;

- the water is re-oxygenated;
- a stable substratum develops;
- slowing the flow allows organic debris to settle and become available as food to benthic invertebrates;
- upstream migration of headcuts is halted and the bed level stabilized;
- locally derived material can be used in construction, thereby keeping costs low.

Generally a stepped series of low structures is preferable to a single large drop. Multiple structures are more stable and provide easier fish passage. They should be placed at different angles across and irregular distances along the riverbed, to offer the greatest diversity of flow strengths and patterns (Nielsen 1996). The structures probably require some form of bank stabilization both upstream and downstream to prevent scour round the edges and possible destruction (Heede 1986, Rutherfurd *et al*. 2000). They also increase hydraulic roughness, and so may reduce conveyance and raise the level of floods (Moses & Morris 1998).

A well-designed grade-control structure will allow fish passage. According to Rutherfurd *et al*. (2000), the following points should be considered:

- mimic natural physical conditions;
- try to establish where the fish generally move and congregate;
- limit areas of free fall;
- keep velocities low;
- limit the downstream slope;
- maintain adequate water depth over the structure so the fish can pass.

Where barriers need to be overcome, rock-ramp fishways, fish bypass channels and culverts can be built. Bypass channels are low gradient earthen or rocky channels that mimic the structure of natural rivers. Rock-ramp fishways mimic the flow conditions in natural riffles (Rutherfurd *et al*. 2000).

The location of these structures within the channel should be designed to work with nature (Luger & Davies 1993). Grade-control structures may fail in high-energy rivers, rivers with excessively high sediment loads (Brookes *et al*. 1996), or in rivers where undermining occurs due to expansion of the scour pool (Rutherfurd *et al*. 2000). Failed structures may become lodged further downstream, redirecting flows and causing new areas of bank erosion. The structures should be maintained so that they do not create scouring and sedimentation problems.

Creating artificial riffles is a common habitat-enhancement technique (Harper *et al*. 1999; Rutherfurd *et al*. 2000). Artificial riffles may be designed that allow fish passage, aid development of upstream pools, act as bed-control structures, re-oxygenate the water and provide habitat and feeding grounds for fish. They also provide intangible benefits through the increased visual and other aesthetic attributes (Petersen *et al*. 1992).

According to Rutherfurd *et al*. (2000) riffle construction could involve recreating a natural riffle formation (a riffle made up of mobile bed material) or a permanent riffle structure. Creating a natural riffle formation involves importing material with a size distribution close to the existing bed material, placing this along the channel at intervals approximating natural riffle spacing, and leaving the final formation to the next flooding events. This is often a temporary solution as the riffle material will gradually be washed downstream and need replacing. Permanent riffles are constructed of tightly packed angular rocks that are larger than those naturally found in that reach. Oversized rocks incorporated into the structure create a complex hydraulic flow, which in combination with the rocks provides a range of habitat conditions. Care should be taken that permanent riffles do not stop aquatic organisms from moving along the channel (Rutherfurd *et al.* 2000).

General design specifications for artificial riffles could include the following.

- Space riffles at between three and ten times the bankfull width (Brookes 1995). The traditional riffle spacing of an average of six times the channel width may require adjustment within this range (Kelln 1994).
- Avoid regular spacing.
- Locate riffles in straight reaches and intervening pools to facilitate a meandering alignment.

Riffles and pools are not usually successfully constructed in ephemeral rivers, in channels with a steep gradient, and where there are severe sediment-transport problems or bank instability (Brookes 1995).

Partial-width structures

Partial-width structures are usually used to protect riverbanks by redirecting flow away from them. Additional effects include (Gore 1985; Brookes *et al.* 1996; Rutherfurd *et al.* 2000):

- realigning the channel;
- removing silt from spawning gravels;
- helping to control water temperatures;
- directing the flow to specific locations such as bank covers;
- increasing water velocities;
- developing a meandering thalweg;
- creating a narrower channel and thereby a deeper low-flow channel;
- creating a scour pool with a depositional area (riffle) downstream;
- formatting bars on which riparian vegetation can be established.

Commonly used partial-width structures are current deflectors, retards, Kellner jacks and submerged vanes (Rutherfurd *et al.* 2000). Current deflectors may be made of metal sheet piling, gabions, rock, boulders, logs or concrete, with the rock ones sometimes called groynes, spurs or dikes (Gore 1995; Brookes *et al.* 1996). They are designed to change the direction of flow and promote scour and deposition, and they can become submerged during floods as their primary purpose is to create habitat and not to protect banks. They usually protrude perpendicularly or at an angle of 45º downstream, although other angles can be used depending on the local conditions. Variations include: a series of deflectors constructed on alternate banks (Brookes *et al.* 1996); upstream pointing deflectors (also called bendway weirs), multiple deflectors and wing deflectors (Rutherfurd *et al.* 2000).

Retards are permeable structures made of a variety of materials such as piles or posts, wire, mesh, cables, tree cuttings, steel and timber. They are mainly used for controlling erosion of riverbanks, stabilising channel alignment and narrowing channels. Traditional retards are a series of piles that extend from the bank toward the centre of the channel. They are constructed on an artificial bench on the outside of eroding bends (Rutherfurd *et al.* 2000). The deposition of sediment behind the bench provides a substratum for plants to become established. The vegetation improves the stability of the bank and restores the riparian appearance of eroded banks (Henderson 1986).

Jacks can be made of wood, metal or concrete; they are employed in areas that are prone to scour, to promote sediment deposition (Rutherfurd *et al*. 2000). Kellner jacks are constructed from three concrete or steel beams, fastened together at right angles at their midpoints. The three beams form the apexes of a triangle that is oriented on the channel bed with two legs upstream and one downstream. By placing assembled jacks in longitudinal rows on the streambed and placing lateral lines of jacks used as tie-backs at intervals, jack fields are constructed. Steel cables can be strung from jack to jack to strengthen the jack field and catch debris (Henderson 1986).

A submerged vane is an instream structure designed to promote bed-scouring by developing secondary circulations in the flow. By manipulating their locations and angles, vanes can be used for bank protection, deepening of channels for navigation, creation of scour pools, reinstatement of meanders, and promotion of depositional zones. They are generally designed to operate best at bankfull flow condition (Stewardson *et al*. 1999) although their use is still limited (Rutherfurd *et al*. 2000). Current information suggests that the angle of the vanes is critical to their success.

Manipulations of substrata and wood debris

With increasing sophistication of rehabilitation works, there is growing recognition that re-creating feeding and spawning habitats is central to the full rehabilitation of rivers. According to Harper *et al*. (1999), the three primary approaches to increasing riverbed heterogeneity in these ways are as follows.

- The river is encouraged to re-create a diversity of substrata and physical conditions by promoting erosional and depositional areas. Partial-width structures, full-width structures and environmentally sensitive river maintenance activities can be employed. This approach is most effective in headwaters.
- Substrata lost in artificial riffles are replaced, and pools re-created. This approach is more suitable for the middle or lower reaches of a river.
- Heterogeneity is created by introducing artificial or natural materials (e.g. quarry rejects, boulders, gravel, woody debris) or by moving the existing bed materials around.

The first and the second approaches have been discussed above. The third approach, which is usually the only one available for the lowest river reaches (Harper *et al*. 1999), is discussed below.

Coarse material such as boulders may be placed in a river to create hydraulic diversity, such as scour pools and downstream bar formations (Rutherfurd *et al*. 2000). Such areas provide cover and substrata for animals, improve aeration of the water, and provide additional habitat for rearing fish. The material can be placed in the middle of a channel randomly or in isolated clusters. If it is placed close to the bank it may cause bank erosion (Brookes *et al*. 1996; Rutherfurd *et al*. 2000), and if the particles are too small they will become covered with the finer bed sediments typical of lower rivers.

Woody debris is an important habitat feature of some lowland rivers, imparting diversity on an otherwise fairly uniform channel, hard surfaces to which organisms can attach and a source of food and refuge for many plant and animal species. Many rivers have lost their source of woody debris through removal of riparian vegetation (perhaps for crops) and of the debris itself (perhaps to meet navigation needs), moves which are now seen as counter-productive in terms of maintaining healthy rivers. The use of woody debris in rehabilitation efforts re-introduces shade, hydraulic diversity and physical habitat for aquatic organisms. Although it does not last as long as rock, it is cheaper and usually more readily available (Brookes *et al*. 1996). It can be re-introduced as cut logs, trunks with attached roots or entire trees (Rutherfurd *et al*. 2000).

Providing cover or shelter

Undercut banks and overhanging vegetation provide shade, shelter against high current speeds and protection against predators for aquatic organisms. Where these have been lost or need to be enhanced, artificial devices can provide additional cover. These devices, fixed to the riverbed or banks, include overhanging platforms, log-over hangs, anchored felled trees, bundles of brush and riprap. Some materials (e.g. branches of trees) are more successful in attracting fish than others, and the spacing and configuration of branches appears to be important as cover requirements differ from species to species and with life stage (Brookes *et al*. 1996; Rutherfurd *et al*. 2000).

8.4.3 Riparian zone and floodplains

See Chapter 9.

8.5 Natural recovery processes

Disturbances of rivers have tended to simplify their structure, straightening channels, eradicating flood channels and smoothing out complex bank structures. After disturbance, rivers tend to revert back to their original condition, but only to the extent that human intervention allows. This is because the same fundamental driving forces that sculptured the original ecosystem, such as climate, the geological formations and topographic features, are usually still intact. This natural recovery, without human intervention, is an alternative option for river rehabilitation (Javela & Jormola 1998). The likely outcomes, at least as far as the channel is concerned, can be evaluated using a model of channel evolution (Harvey & Watson 1986). This qualitatively describes the stages of recovery of a degraded river from total disequilibrium to a new state of dynamic equilibrium, and may be used to identify reaches that are more amenable to natural rehabilitation. However, such qualitative descriptions of channel recovery have limited use for the determination of recovery time scales and what is likely to happen over any specific period.

Natural recovery can be charted using indicators such as (Bartley & Rutherfurd 1999):
- the re-appearance of pools and riffles;
- development of a meandering thalweg;
- changes in bank slope and development of flood terraces;
- adjustment and stabilisation of channel cross-sections;
- increased variability of flow patterns;
- armouring of heterogeneous substrata;
- establishment of aquatic and riparian vegetation;
- sorting of bed substrata.

Limitations to this approach are imparted by such factors as urban development, regulated flow regimes, insufficient stream power and landownership. Furthermore, irreversible changes such as the blasting of bedrock may mean that natural recovery within any practical timescale is impossible (Jarvela & Jormola 1998).

8.6 Summary

Successful rehabilitation efforts are based on three main activities or measures. These are environmentally sensitive river maintenance activities, soil bioengineering or soft biotechnical engineering practices and measures to mitigate the effects of hard engineering practices. Environmentally sensitive measures include: reducing the frequency of maintenance activities; modifying only one bank; reducing dredging operations; careful disposal of dredged material; controlling stock access; reducing the frequency and intensity of mowing; maintaining riparian and aquatic vegetation; controlling weeds; and using sediment traps. New innovative soil bioengineering or soft biotechnical engineering techniques, such as bank protection using geotextiles and terraced geocells, provide a more natural appearance to rivers than hard engineering practices. These have many limitations, however, and are often best used in conjunction with hard engineering rehabilitation tools such as riprap and gabions. Natural recovery of the river channel, without intervention, is an option where time and space allow and where other limitations, such as excessive water abstraction, are not present.

9. REHABILITATION OF BANKS, RIPARIAN ZONES AND FLOODPLAINS

9.1 Introduction

Channelisation, especially straightening, may result in the loss of meanders (Petersen *et al*. 1992), in bank erosion (Brookes 1990), and in disconnection of the floodplain from the river channel (Gore & Shields 1995). Land-use changes, such as conversion of natural riparian vegetation into agricultural fields, result in fragmentation or total loss of the riparian zone. In this Chapter, methods used to rehabilitate banks, riparian zones and floodplains are outlined, together with confounding issues due to, for instance, human settlements.

9.2 Re-connecting the floodplain with the river

Zsuffa & Bogardi (1995) described floodplains, because of their high ecological potential, as core function areas for the rehabilitation of the whole degraded natural environment. The ecological problems associated with floodplains of regulated rivers can be classified into two groups: those linked to land use and those linked to an altered water regime. The first group comprises habitat destruction, disturbance and pollution, caused by agriculture, forestry, mining and recreational activities. Terminating these activities or reducing their intensity to a less damaging level can help solve the problems.

Three main eco-hydrological problems in floodplains arise from altered water regimes. First, there can be increased desiccation because of lowered river levels. Water levels in the River Rhine in the Netherlands and in the Danube in Hungary have dropped by 1.5 m during the past 90 years (Zsuffa & Bogardi 1995). Shorter inundation periods for floodplains and lowered groundwater levels led to a shift from alluvial wet ecological units towards drier ones. Second, channelisation can lead to degradation of fish habitats, because of shrinkage of floodplain water bodies and fast water-level fluctuations. Rapid fluctuations can result in deposited eggs and larvae becoming stranded and dying. Third, increased nutrient concentrations from point and non-point effluent sources can lead to eutrophication and algal blooms, with the situation exacerbated by desiccation and the ensuing lack of dilution capacity of the water.

9.2.1 Restoring flows in remnant channels within the floodplain

There are two basic methods for restoring connections between the floodplain and river channel: to restore flows in remnant channels within the floodplain, or to create secondary channels that allow new interaction between the river and the floodplain. Another possibility, to simply take down embankments so the floodplain can flood all along the river, is not viable if the floodplain area has been developed extensively.

The use of remnant channels is demonstrated using the example of the Kissimmee River demonstration project, described by Toth (1993) and Toth *et al.* (1998). The river was channelised between 1962 and 1970 and transformed into a series of impounded dams. This altered the hydrologic characteristics of the system, and eliminated the wetland, fish and wildlife values of the river and floodplain. In 1971, post-channelisation hydrologic manipulations began, to explore the potential for lessening the loss of wet prairie. The objective was to encourage re-establishment of lost plant species through seasonal draw-downs of water levels in several of the channelised pools.

The above-mentioned project was conducted in a 19.5 km long section of remnant river and floodplain between two weirs S-65A and S-65B in the Osceola, Okeechobee and Highlands Counties. The project had four components: 1) implementation of a pool-stage fluctuation schedule; 2) construction of three weirs; 3) creation of a "flow-through" marsh, and 4) hydrological and hydraulic modelling studies. The objective was to restore the pre-channelisation flow regime in terms of seasonal flows, average in-channel flow velocities, over-bank flooding frequencies, and over-bank stage and recession rates (Brown *et al.* 1997). The loss and degradation of wetland habitat that had resulted from adherence to stable pool stages was countered by the pool-stage fluctuations. An annual 11.9 to 12.8 m water level fluctuation schedule was designed to re-establish seasonal water level fluctuations over about 1080 ha of floodplain, including 526 ha that had been drained since channelisation. The weirs stimulated backfilling of the channelised river by diverting flow through adjacent remnant river channels and providing additional inundation of the floodplain. The resulting river flow regime and flooding patterns were a function of upper-basin discharge characteristics and the flow-diversion efficiency of the weirs. The project illustrated that the re-introduction of flow and associated fluvial processes in remnant river channels enhanced the diversity and quality of degraded river habitat. There was a rapid rehabilitation of river and floodplain wetland vegetation and also the early stages of re-establishment of natural river species. Annuals and perennials re-colonised exposed riverbanks and other areas where vegetation was removed. The hydrological manipulations failed, though, to change the plant community in undisturbed wetlands where dense stands of dominant plants established before the draw-downs prevented the establishment of other plant species.

Rehabilitation was also attempted of the Southern Gemenc floodplain along the Danube in south Hungary (Zsuffa & Bogardi 1995). The area contained three cut-off meanders (oxbow lakes) of the river that were connected to the river and each other by means of small channels called foks. The cut-offs filled and drained through the foks as flow fluctuated in the Danube. Channelisation resulted in channel bed erosion, leading to decreased water levels, shorter inundation durations, decreased groundwater levels and rapid water level fluctuations. As a result, the floodplain became exposed to desiccation. Eutrophication was caused by pollution from land uses and exacerbated by decreased water levels. Rehabilitation was attempted because it was recognised that floodplains can function as core areas for rehabilitation of the whole degraded natural environment due to their high ecological potential. Restoring the water regime could be achieved by controlling the flow in the links (fok channels) connecting the different floodplain depressions to the river and to each other. Re-linking the various units of the system to each other could be done in two ways: passively, by creating channels and weirs of an appropriate size; or actively, by operating sluices built into the links. These water-control measures became a powerful means to counteract desiccation and to respond to the specific hydrological needs of various species. The passive way was chosen, but whether or not the approach was successful was not reported.

A Missouri River chute (side channel or oxbow lake) was isolated due to channelisation (Harberg *et al.* undated). The channelisation works resulted in the loss of water-surface area and a corresponding loss of physical-habitat features such as chutes and sloughs that provided spawning and rearing habitat for native fish species, feeding and nesting habitat for shore birds, and migration routes. Rehabilitation involved restoring conditions similar to those present before channelisation, by re-connecting the upstream and downstream ends of the chutes to the river and allowing chutes to develop their historic patterns and cross-sections. Post-project monitoring began in May 1993. Early observations showed the presence of shovelnose sturgeons (a species adapted to lotic environments) in the chutes. Erosional and depositional zones and snag habitats had also formed.

9.2.2 Creating new secondary channels

Excavation of new secondary channels in floodplains enhances interaction between them and the main channel. This approach was used to restore some ecological integrity to rivers in The Netherlands, where the main channels are relatively deep, narrow and fast flowing. The 'summer' channels are bordered on both sides by summer dykes (embankments or levees) that keep the normal flow of the rivers out of their floodplains. Winter dykes, or high ground, mark the outer edge of the floodplains. Secondary channels were proposed, to provide living conditions for riverine organisms that had disappeared from the main channel. The shallower, slower flow in the secondary channels facilitated processes such as sedimentation, growth of vegetation and afforestation, and provided habitats not available in the main channel (Schropp & Bakker 1998). Since the project was not completed at the time of publication, its success could not be reported on.

The stability and sediment regimes of secondary channels can cause concern, and need to be factored into any such rehabilitation programme. There can be horizontal instability through meandering, and vertical instability through erosion and deposition. Horizontal instability is determined by the degree of bank erosion, which occurs mainly on the outer bends where the flow is faster. Repeated overbank deposition during floods along low gradient meandering rivers often leads to thick accumulations of fine-grained material (clay and silt) on their floodplains. Removing the upper, resistant clay mattress may result in instability if, during excavation, the level of the floodplain is lowered through lateral shifting of meanders (Barneveld *et al.* 1994). Vertical instability can arise from either deposition or erosion. Lower velocities and thus carrying capacities for sediments in the secondary channels result in deposition (Schropp & Bakker 1998), and perhaps eventual complete siltation of the channels (Schropp 1995). This problem can be alleviated by building sediment traps at the entrance to secondary channels, to reduce flow of sediments into them (Barneveld *et al.* 1994; Schropp 1995; Schropp & Bakker 1998). The problem can also be addressed with regular dredging, but this will detrimentally affect development of habitats and species in the secondary channel. Erosion of secondary channels can occur if their sediment-carrying capacity is greater than the sediment inflow (Barneveld *et al*. 1994).

9.3 Re-instating meanders

Meanders were re-instated in the channelised River Gels Å (Iversen *et al.* 1993), Pine River (Newbury 1995), Whittle Brook and River Alt (Nolan & Guthrie 1998), Rivers Brede, Cole and Skerne (Vivash *et al.* 1998) and River Skerne (Eden *et al*. 1999). In most cases the objective was to restore sinuosity and connections to the floodplain. The re-created meanders were located along old meandering courses of the rivers with guidance from old maps. Factors taken into account included width, depth and discharge capacity of the new reaches, pool and riffle spacing and preferred fish habitat.

The general rule of thumb for re-instating meanders is to set them five to seven stream-widths apart or approximately one-half of the meander wavelength. Meander wavelength is normally about 8-10 times the width of the channel. This general rule may not be applicable to all stream types, and it is preferable to develop relationships for a region by studying meander patterns on aerial photographs of undisturbed sections. Meander spacing is normally very variable, and this should be worked into the design by taking into account the location of trees and boulders, and variations in soil or substrata. Hasfurther (1985, cited in Gordon *et al.* 1992) summarised four approaches to meander design.

- *The carbon-copy method* The meander is constructed to be exactly the same as the pre-disturbance form, or to follow the pattern of a similar reach in an undisturbed section. This method assumes that other factors affecting stream patterns (e.g. discharge bed materials) have remained the same.
- *Empirical relationships* Specifications for meanders are developed for specific regions and/or stream sites and only applied to those areas.
- *The "Natural" approach* The stream is allowed to seek its own path. A serious disadvantage of this approach is that the stream may take a long time to reach a stable form, with high erosion rates and sediment movement in the meantime.
- *A "systems" approach* Advocated by Hasfurther, this approach includes an analysis of undisturbed meanders, an evaluation of the geomorphology of the disturbed area and consideration of the interaction between the stream and the surrounding areas (Gordon *et al.* 1992).

Designing new meanders on straightened reaches can pose several problems related to the unavailability of space. Re-instating meanders was constrained on the River Skerne by the existence of drains, a gas main and electricity cables that restricted the size and location of the meanders (Eden *et al.* 1999; Vivash *et al.* 1998). A small ox-bow feature was designed for Whittle Brook that appeared feasible on paper, but was confounded by the need to maintain machine access to the flood bank (Nolan & Guthrie 1998). Where channels cannot be re-meandered due to space limitations, flows can be diversified and riverside plants encouraged by in-channel reflectors (Eden *et al.* 1999; Iversen *et al.* 1993). Re-instating meanders is usually not done in isolation, but goes hand in hand with bank reshaping, bank stabilisation and revegetation practices (Vivash *et al.* 1998).

9.4 Stabilising stream banks

Stabilisation of stream banks is important in terms of controlling sediment inputs to the stream. Heavy sediment loads blanket river beds, smothering habitats, spawning grounds and food sources, and in-filling vital refuge areas in interstitial spaces between river-bed cobbles. The loads also may reduce the life of downstream in-channel dam reservoirs, threatening the viability of major water-resource developments. Banks may be stabilised by: 1) the use of vegetation alone, 2) vegetation with structural control, 3) vegetation, structural control and bank shaping, (4) structural control alone and (5) bioengineering methods. This topic has been covered in Chapter 8, and is expanded here through case studies and some discussion of factors to consider when employing each approach.

9.4.1 The use of vegetation

Case studies

Hotopia Creek in northwest Mississippi, U.S.A., used to have an incised channel due to channelisation between 1840 and 1965. Rehabilitation was attempted because aquatic habitats were severely degraded, with riparian vegetation, woody debris and pool habitat in short supply. Woody plantings and stone were used to accelerate the rehabilitation of aquatic habitat. In 1991, in a pilot study, dormant willow posts 0.09-0.13 m in diameter and stakes 0.03-0.04 m in diameter were planted 0.0-1.9 m apart along the base-flow channel. (Shields *et al.* 1993). Native black willow *Salix nigra* was used because of properties such as flood tolerance, resistance to insects and disease, relatively high survival rates when planted as dormant cuttings, rapid growth, local availability and prolific root development. Furthermore, several small individuals of this species were already growing at the study site, indicating that habitat conditions were suitable. Two-thirds of the posts and stakes were washed away due to flooding after the planting, but those that remained and survived developed a lush growth of branches and leaves.

Marginal vegetation species were reported as particularly useful for protecting eroding banks from the force of waves generated by boat traffic in British waterways (Brookes & Hanbury 1990). *Phragmites australis* (common reed), *Schoenoplectus lacustris* (bulrush), *Typha angustifolia* (reed mace) and *Acorus calamus* (sweet flag) were the four main species used on the River Thames. *Carex riparia* (greater pond sedge) and *Carex aculiformis* (lesser pond sedge) were preferred for canals with low to intermediate levels of boat traffic, because they form dense continuous marginal fringes that do not encroach on the water channel beyond a water depth of approximately 0.20 m. All these plants are capable of absorbing the wave energy from boat wash, maintaining bank stability, preventing scour of the substratum and facilitating the deposition of sediment.

Factors to consider

McCann & Lindley (1998) advised farmers how to rehabilitate eroded riverbanks and wetlands. They pointed out that herbaceous plants have rapid spreading capabilities, dense near-surface root mats, and provide good surface cover of the soil. The plants absorb fast flowing water rather than deflecting it, inducing sediment deposition around their stems and so tending to raise the floor of the channel. They are extremely effective in combating scouring, and enhance the stability of gentle or shallow slopes, but are not effective as stabilisers of steep banks due to their shallow rooting depth. Trees contribute to the cohesion and stability of steep banks, providing that the roots reach as far down as full bank height and the toehold and bank face are protected from undercutting. Species with vigorous rooting characteristics should be chosen, and their stabilising effect will accelerate natural plant succession (McCann & Lindley 1998). Native species of local stream banks should be used, since although alien species may have higher survival rates (Henderson 1986; Gordon *et al.* 1992; McCann & Lindley 1998) they may be unsuitable bank stabilisers. Native species have evolved to cope with local conditions and will best support local wildlife. Among the native species, some species will be of more value than others as food or habitat for fish, invertebrates, waterfowl and other wildlife.

Henderson (1986) summarised the criteria for selecting native species. They should be able to: 1) withstand the degree of anticipated inundation, 2) provide year-round protection to banks, 3) have the capacity to become well-established under sometimes adverse soil conditions, 4) have root, stem and branch systems capable of resisting erosive flows.

Understanding the influence of vegetation on water levels and limitations to its effectiveness in rehabilitation, aids stream management. In general, increasing the vegetation cover will increase the hydraulic roughness of and friction in the channel, so water will move more slowly. Water levels will be higher for a given discharge, and so flood levels could increase. This is an important consideration when designing flood-management strategies (Eley & Keller 1999). Additionally, silt deposition in the vegetation may be desirable from an ecological point of view, but the build-up can also lead to local flooding (Gordon *et al.* 1992). The use of vegetation for bank stabilisation was completely banned at certain times in Central Europe because of the expected loss of flood capacity due to growing plants. Finally various other processes may limit the effectiveness of vegetation as a stabilising agent. In addition to their differing abilities to withstand flow stress, many plant species will not grow in permanent water of certain depths (Jaeggi 1989). Moving bed loads of coarse material can also abrade the plants, reducing their growth and strength.

9.4.2 Vegetation and structural control

Replanting stream banks with native vegetation, especially shade-producing woody vegetation, is often central to environmental rehabilitation efforts. However, high flows are likely to remove plantings installed along the margin of actively eroding streams before they can become established. Planting is thus often done in conjunction with active measures (e.g. use of revetments) to increase the resistance of stream banks. Revetments are any type of material, but usually stones or concrete, laid on the bank surface to protect the bank from erosion. Section 8.4 gives further information on revetments. Designs featuring stone revetments for lower bank protection, and various types of vegetation for upper bank protection, are growing in popularity (Gore & Shields 1995; Moses & Morris 1998), because they are preferred from an environmental point of view.

Vegetation with toe protection at the base of the bank was used in three experiments in the Yazoo River basin, Mississippi, USA (Shields *et al.* 1995b). Arrays of concrete jacks laced together with steel cables were used to protect the toe of the bank. One site, consisting of a sinuous reach with concave banks, was protected by placing a row of stone along the bank toe. As part of the experiment, water-elm seedlings were planted landward of the toe for protection on every other bend. The entire bank toe remained stable in spite of a near bankfull discharge 11 days after construction, with no significant damage to toe or plantings.

Powell and Washington Parks, Waukegan River, Illinois, USA, suffered from bank erosion due to high velocity floodwaters that destroyed park bridges and city sewer lines. The lack of instream habitat and destruction of park property were key factors in selecting a Lunker installation (Section 8.4) (Roseboom 1994). Vegetation introduced in the soil placed over the lunkers grew over the eroded areas within two weeks. This technique together with upper bank vegetation proved to be very durable, and provided streambank stabilisation when floodwaters later overflowed the entire park.

Martin Dale Creek (MDC) and Martin Dale Tributary (MDT) drain adjacent catchments in the upper Yazoo River basin, Mississippi, U.S.A. (Shields *et al.* 1997). They used to be incised streams with eroding banks due to channelisation. A metal-sheet piling weir with a stone-protected approach channel and stifling basin drop structure was constructed on MDC. A stone weir was constructed in MDT, and a ridge of stone was placed along the toe of a total of 700 m of concave banks upstream of the weir. About 3000 willow sprouts were planted landward of the stone toe along MDT, while 4282 dormant willow posts were planted in the face and along the toe for 500 m of both banks of the MDC immediately upstream of the drop structure. The use and correct placement of toe rock are critical for long-term bank stabilisation, as this is the key area that protects and supports the upper bank. The channels were laterally stable during periods between channel surveys. Water depths, widths and pool habitats increased after construction. The pools developed immediately upstream of the weir and adjacent to the stone toe. Richness of fish species increased, with the species composition shifting away from species typical of shallow, sandy runs toward pool dwellers. Physical and biological responses were more persistent for the stream treated with the stone weir and toe protection.

Rocks of diameter 1.2 - 1.5 m were laid in solid ground for the realignment of Koonung Creek, Australia (Shaw 1999). Different bank pressure points were protected with different weight rock, to protect the bank as the creek re-established its own equilibrium during periods of high flows. Placement of the rock was staggered to maximise the points of contact and improve interlocking. Surface heights between rocks varied, to enhance the overall natural appearance. Voids in and around the rocks were in-filled to ensure long-term structural integrity, using small material collected from the floor of the quarry after blasting the bigger material. Aquatic macrophytes, sedges, herbs and woody shrubs and trees were established in amongst the rockwork to soften the appearance. The project was successful with the rockwork providing erosion control measure and subsequent bank stability and the vegetation providing habitat and an aesthetically pleasing appearance.

9.4.3 Vegetation, structural control and bank shaping

The use of erosion control fabrics and re-seeding with quick-starting groundcover plants is a common measure to ensure initial stability of re-graded banks until the planted vegetation becomes fully established. Establishing vegetation on re-graded stream banks has its own difficulties, including the loss of upper soil layers on banks excavated to a gentler slope. The re-graded banks formed from subsoil (thus poor in topsoil or organic matter), may be a poor medium for plant growth and may need nutritional additions (Moses & Morris 1998).

In a rehabilitation project of the Elbæk River, Denmark, bank slopes were re-graded so that they did not exceed the angle of repose, that is the stable angle of the bank material (Brookes 1990). Existing trees were retained to provide some bank stability, and a further 500 willows and birch planted to enhance long-term stability. Exposed earth surfaces were reseeded with species of grass, whilst crossover points between the new and old channels were protected with gravels and riprap. The project appears to have been successful, but it is too early to assess its long-term success. A procedure for post-project assessment has been established.

Three reaches along concave banks of the Yazoo River basin, Mississippi, U.S.A. were stabilised using the same method (Shields *et al*. 1995b). Shaped banks were graded so that the top of the finished bank sloped away from the channel to prevent drainage of runoff back down the face of the slope. Study reaches with shaped banks were subjected to severe climatic stresses during the 12-year period of observation, but they remained well vegetated and stable. Planted and naturally introduced native species gradually invaded areas protected by rock and concrete blocks, colonizing deposited sediments and interstitial soils. This helped increase the strength of the block revetments.

9.4.4 Structural control

The Sandton Municipality, Gauteng, South Africa, used Waterlöffel interlocking concrete blocks (Section 8.3) to stabilise an embankment that had eroded by more than 3 m vertically during a summer storm (Municipal Engineer 1992). The Waterlöffel can accommodate stream velocities in excess of 10 m s^{-1} and also creates a turbulent boundary layer that, unlike reinforced concrete, assists in reducing flow velocity.

9.4.5 Bioengineering methods

Soil bioengineering entails combining biological and engineering concepts and methods to control stream bank erosion. Rehabilitation techniques employing this approach usually use a mixture of natural wood and stone, living vegetation and man-made materials. Bioengineering systems involve placing materials such as geotextile fabric or mesh, terraced geocells, interlocking concrete blocks, fabric-encapsulated soil (FES), stone, rock, wood or vegetation in contact with the bank. Section 8.3 outlines the different bioengineering methods.

Rapidly fluctuating river levels and scour in the Little Miami River, Hamilton County, USA, eroded high riverbanks to nearly vertical slopes (Hoitsma 1999). Terraced geocells over a stone foundation were used to stabilise the lower portion of the bank. FES was used to stabilise the mid bank. The results were positive. There was a small adjustment to the rock toe foundation and minor settling, but the bank was stable after a few high flow events.

Modern geotextiles were also used in British canals, to stabilize and protect banks from erosion and to hold young plants in place until they became established (Brookes & Hanbury 1990). Using an approach called “reinforced vegetative bank protection”, eroding banks were reshaped with dredged material. Geotextile fabric was pinned to these with wooden stakes. Pockets within the fabric were filled with locally available plant material during spring and early summer. A healthy reed fringe was expected to develop by the end of summer, completely concealing the fabric. The fabric has an indefinite life and will remain *in situ* to protect the bank should the vegetation be damaged by grazing, trampling or cutting.

Some biotechnical or soil bioengineering bank stabilisation techniques, (for example vegetated geogrid and those described in the preceding examples), work well along high-energy streams. This is because the fabric will still protect the soil from erosion even if the vegetation is damaged. Plant-based techniques, such as contour fascines (bundles of willow stems) have also been used in such situations, but they frequently wash out and actually exacerbate erosion. Plant-based techniques may also be inappropriate where bank material is non-cohesive.

9.5 Riparian zone rehabilitation

Odum (1981, cited in Terrill, 1999) described the riparian zone as an “interface between man’s most vital resource, namely water, and his living space, the land”. Rehabilitation of the riparian zone involves revegetation, perhaps together with setting land aside for this purpose. The role required from the riparian zone will dictate the species chosen (Petersen *et al*. 1992).

A healthy zone of riparian vegetation improves water quality by trapping sediment, nutrients and other contaminants; reduces rates of bank erosion and downstream flooding through protecting the banks; provides shade and so reduces temperature levels and primary productivity in the river; provides food and diverse habitat for aquatic and riparian animals; provides refuge and a migratory corridor for wildlife; and creates an aesthetically pleasing landscape area for recreation (Prosser 1999; Terrill 1999).

Several constraints need to be taken into account if riparian zone rehabilitation is to be successful. Carr *et al.* (1999) outlined these as follows.

- Re-vegetation may not always be the best and only erosion control measure. Some structural work may be needed first. Re-vegetation is often seen as a cheap alternative, but it can be expensive where the causes of erosion have not been fully assessed first.
- Older more established plants can prevent the establishment of more desirable species. Understanding the condition and succession stage of existing vegetation, and its likely effects on new plants, is vital for successful re-vegetation.
- All plant species have optimum conditions for growth where they will establish themselves most vigorously. Some species naturally occur higher up banks than others, and some tend to be in either sun or shade. Species planted in inappropriate locations will not thrive, threatening the success of the project.
- Plant species occur at different stages of succession in plant communities. Some species appear first, as robust colonisers, whilst others naturally appear later in more established vegetation. Species should be selected for the desired function and location with this in mind.
- If the site is not prepared well, non-preferred or weedy species might out compete the desired ones. Site preparation should be done to provide the new plantings with a competitive edge.
- A single planting method for all sites is inappropriate. Sites vary in site conditions, natural plant communities and remnant vegetation, and the planting design should take all these into account.

Carr *et al.* (1999) and Webb & Erskine (1999) outlined the data needed for a successful riparian zone rehabilitation project.

- *Vegetation history* Historical photographs and accounts of vegetation can be used to determine the type of vegetation found at the site before the land use changed. This should be done with caution as this type of information can be qualitative, incomplete or inaccurate. Adjacent areas can be surveyed to provide information on plant species and communities suitable for the rehabilitation area. This provides a reliable method for selecting appropriate species for planting, since it will be possible to construct a vegetation template (design) for the area. Properly supervised and documented trial plantings can be used if there are no historical data or suitable adjacent areas.
- *Site characteristics* An understanding of the links between vegetation and the physiography, geomorphology, aspect, soils and substrata down the slopes is vital. Riparian species are influenced by all these factors and each will grow most vigorously in an environment that best suits it.
- *Vertical plant zonation* Saturation levels vary up the riverbank, and different species tolerate wetter soils and more inundation than others. The natural location of each species should be known and followed in plantings.

- *Flood regimes* Floods of different frequency, magnitude and duration reach to various elevations above the streambed. New plantings are very vulnerable to flood disturbance and may be scoured out by even the smallest flood. Timing is therefore critical, and an understanding of rainfall patterns and stream flow dynamics should guide when plants are introduced to allow them the greatest chance of becoming established.
- *Planting method* Site conditions and the species selected will determine the planting method used. Standard tube-stock planting entails the planting of seedlings where water-lancing jets are used to create narrow cavities in which the seedling is placed. Only the top 30 to 50 cm of the seedling is left protruding above the soil surface following infilling of the planting cavity. The main disadvantage of this technique is that the seedlings need protection from grazing, direct sunlight and/or frost damage. Direct seeding can be used where large areas need to be revegetated. Long-stem planting involves first growing standard tube stock to a height of approximately 1.5 m, whilst maintaining a relatively small root mass. These long-stem seedlings can then be planted to depths of up to 1 m. The long stems provide protection to the roots from direct sunlight, and are less likely to be washed away during minor flood events than standard tubestock seedlings. The above methods can be used together (Carr *et al.* 1999; Webb & Erskine 1999). Hicks *et al.* (1999) compared growth and survival rates of long-stem tube stock and standard tube stock of four species, namely: *Callistemon viminalis*, *Tristaniopsis laurina*, *Waterhousea floribunda* and *Casuarina cunninghamiana* at two sites: Seaham weir on the Williams River and Marlee on Dingo Creek. At both sites, long-stem tube stock of both species had a significantly higher growth rate than did standard tube stock. It was suggested that this occurred because the long-stem plants had access to subsurface moisture. Assessment of root development in some of the long-stem specimens of *Eucalyptus camaldulensis* revealed root growth from growth nodes along the full length of the buried stem.

9.6 Riparian buffer strips

The riparian zone is sometimes referred to as a buffer strip because of its location and functioning at the interface between terrestrial and aquatic ecosystems. The buffer strip stabilizes the banks along the stream while creating complex habitats that trap and retain water, sediments and nutrients moving down toward the river from the landscape (Petersen *et al.* 1992). In the Johnstone River catchment, Queensland, Australia, the majority of sediment eroded from agricultural land was not stored within the catchment, but was quickly delivered to the river mouth (Prosser 1999). Improved sediment trapping in the riparian buffer strips provides one of the few opportunities to stop eroded soil from leaving the catchment without clogging the streams, but riparian buffer strips were not put in place to minimise sediment delivery to the river.

Buffer strips are very effective in agricultural and forestry areas for reducing sediment inputs to rivers and improving the quality of the water draining off the landscape (e.g Newbold *et al.* (1980); Petersen *et al.* (1992); Line *et al.* (1998); Karssies & Prosser (1999); Line *et al.* (1999); McKergow *et al.* (1999)). Identifying the conditions within the catchment that lead to the greatest delivery of sediments to the river environment allow appropriate buffer zones to be planned and maintained. For example, two types of buffer strips, grass and rainforest, were studied in Queensland, Australia (McKergow *et al.* 1999). Grass cover trapped more than 80% of the mobile sediments, which were defined as water-stable soil aggregates of clay and silt predominantly 2-4 mm in diameter, thus providing the longest residence time. High delivery to the river from grassy areas occurred only on steeper slopes with little grass cover. The remnant rainforest acted as a temporary store, but the sediment easily moved on to the stream.

In a laboratory experiment, Pearce *et al.* (1997, cited in Line *et al.* 1998) illustrated that the length of a vegetated buffer of Kentucky bluegrass was more important for filtering sediments than the height of the grass. Cole *et al.* (1997, cited in Line *et al.* 1998) found that Bermuda grass buffers were effective at reducing pesticides and nutrient runoff from golf courses, regardless of the height of the grass or of aerification treatments (Line *et al.* 1998).

Line *et al.* (1998) reviewed the role of buffer zones in reducing non-point source pollution to rivers. They found that when runoff from poultry litter was applied to fescue grass, concentrations of metals decreased significantly with increasing distance across the vegetated buffer length. Overland flows through Bermuda grass and annual rye grass also effectively removed phosphorus in runoff. Line *et al.* (1999) reported that buffer strips were effective in reducing wheat fertilizer runoff, dairy loafing lot runoff, beef cattle pasture runoff, swine lagoon waste-water runoff and poultry waste runoff.

9.6.1 Dimension of buffer strips

The effectiveness of buffer strips in trapping sediment and reducing pollutants entering the streams is usually strongly related to their dimensions (Newbold *et al.* 1980). Karssies & Prosser (1999) suggested that a minimum filter width in areas where soil erosion is occurring should be 2 m, with this width increasing as erosion increases. Rates of soil loss are related to slope, soil erodibility, soil erosivity and the amount of time that the soil is bare. Patty *et al.* (1997, cited in Line *et al.* 1998) found that 6-18 m width buffer strips reduced suspended solids by 87–100%, lindance by 72-100%, altrazine and its metabolites by 44-100%, soluble phosphorus by 22-89% and nitrate by 47-100%.

Newbold *et al.* (1980) studied the effects of logging on macro-invertebrates in streams without buffer strips, with narrow (<30 m) strips or with wide (>30 m) strips. Macro-invertebrates in stream with wide strips showed no effect from the logging, while the streams with no strips showed significantly lower diversities, but high densities of robust taxa such as *Baetis*, *Nemora* and Chironomidae. Logging had a significant impact on the streams with narrow strips, but the exact significance probability could not be determined because two blocks had one control site in common. Wider buffer strips become increasingly effective, whilst also providing good habitat and refuge for wildlife. Peterjohn & Correl (1984, cited in Petersen *et al*. 1992) reported that a 50 m width riparian forest separating agricultural fields from a stream removed 89% of nitrogen and 80% of the phosphorus draining into it. It was estimated that there was a net removal of 11 kg ha yr^{-1} of organic nitrogen, 47.2 kg ha yr^{-1} of nitrate nitrogen and 3kg ha yr^{-1} of particulate phosphorus.

9.7 Constructing wetlands to manage water quality

In many parts of the world, agricultural lands have been created from wetlands by draining the land and exploiting the rich soils. Ditches may have been dug or drainage tiles laid under the fields to encourage drainage. Run-off from these areas fed fertilisers and pesticides directly into watercourses. These nonpoint source pollutants can be controlled through mini-wetlands created at a point before the drainage enters the stream. Such wetlands act as natural ones, attenuating floods and improving water quality by trapping the pollutants. Wetlands of gravel, cattail, common reed and spike rush, for instance, removed 39% of orthophosphates, 46% of suspended solids and 50% of dissolved copper from storm water in a detention pond (Line *et al.* 1998).

Petersen *et al.* (1992) described small horseshoe wetlands at the end of drain tile systems. Semi-circular-shaped excavations in the last 8 m of the drains allowed the water they were carrying to flow over a grassy, shrub section before entering the stream. Nitrogen and phosphorus were extracted from the water by each wetland. Line *et al.* (1998) described constructed wetlands in the Czech Republic, Canada, the U.S.A. and New Zealand that showed various levels of success in purifying waters.

Principles to be incorporated into the design of constructed wetlands were suggested by Garner (1997).

- Install sediment traps in the form of a series of weirs or separate small dams.
- Plant reedbeds to trap suspended solids and help purify the in-flowing water.
- Use of aerators, or a series of waterfalls and cascades, to introduce air into the water.
- Install litter traps to collect rubbish discharged through the storm water system.

9.8 Summary

This chapter accesses literature on a range of off-channel rehabilitation activities, such as reconnection of floodplains to rivers, re-instatement of meanders, stabilisation of stream banks, and rehabilitation of riparian zones. It also discusses the use of riparian zones as buffer strips and artificial wetlands to manage water quality.

The riparian zone/corridor may be defined as the interface or transition area between the active channel and the terrestrial landscape, and occurs adjacent to the channel. It is located within the macro-channel, that wider area outside the active channel that is inundated at intervals greater than about one year. Floodplains also form part of the interface, and can be situated at various positions with flooding frequencies ranging from several times a year to once every few years. Human activities have resulted in the fragmentation and narrowing of these areas, with reduction in their complexity and connectivity with the river, and a resulting decline in the ecological health and productivity of the whole river ecosystem. Because of the high value of land, and of past planning practices and limited foresight by earlier generations, there may no longer be space available along most rivers for re-instatement of floodplains or wide riparian zones.

Floodplains can be re-connected to river channels using two main methods: reconstruct relict secondary channels or create new ones. Meanders are usually re-instated on old meander paths, but this may not be possible where human developments have covered these. Five methods of stabilising stream banks are given along with their benefits and problems. These are: 1) the use of vegetation alone; 2) vegetation with structural control; 3) vegetation and structural control with bank shaping; 4) structural control alone and 5) bioengineering methods. Rehabilitation of riparian zones and use of them as buffer strips requires a good understanding of environmental conditions at the site, of what is required from the rehabilitation, and of the biology of any plants that might be used, so that the right species are used in the right places. Well designed plans are essential for success in both rehabilitation activities and the construction of artificial wetlands.

10. REHABILITATION: IN THE CHANNEL

10.1 Introduction

In urban situations where riparian space is limited, increasing the diversity of instream habitat may be the only rehabilitation option available to the stream manager. This has advantages in that it is often easier to obtain permission to work within the boundaries of a river channel than to effect changes in the riparian corridor and the greater catchment (Fogg & Wells 1998). Ideally of course, restoration of a fully functional stream corridor should be aimed for within a well-managed catchment and riparian zone, since man-made instream structures are rarely sustainable in the long term (Fogg & Wells 1998). In this Chapter, the following are discussed: goals for instream rehabilitation (Section 10.2), relevant hydraulic and geomorphological processes (Section 10.3), rehabilitation of rivers with dams (Section 10.4) and the range of options for instream structures (Section 10.5).

10.2 Goals for instream rehabilitation

Interventions within the boundaries of the channel are usually aimed at enhancing diversity of hydraulic, substratum and other environmental conditions, by introducing structures that create habitat diversity and cause variations in current speed (Rutherfurd *et al*. 2000). Such rehabilitation efforts could be successful if simplification of instream habitat is the limiting factor affecting recovery at the site (Walsh & Breen 1999). If, however, other limiting factors are present, such as poor water quality, they may achieve little if done in isolation. Rehabilitation of physical habitat is most often attempted by altering the range of velocities, depths, substrata, and even the amount of incident light, in the targetted reach. Instream rehabilitation techniques are most often applied to channelised rivers in agricultural areas or canalised urban rivers, since such systems often have a very uniform channel. In most cases, the aim is to re-introduce or mimic natural conditions to the highest extent possible.

10.3 Designing instream structures

Where understanding about streams and their dynamic nature is poor, structures installed in channels to enhance diversity and naturalness may fail or have limited life-spans. In a 1995 survey, 1234 instream structures were evaluated for their general effectiveness, habitat quality and use of the structures by fish (Fogg & Wells 1998). Three major findings were reported:

- generally, instream structures provided an increase in available fish habitat;
- one fifth of the structures required regular maintenance;
- in conditions of inadequate flows and excessive sediment delivery, the structures had limited life-spans and were short-lived in terms of habitat improvement.

Thus, without an understanding of the desired result and the channel processes that will work with or against the rehabilitation effort, there is a risk that channel rehabilition will be unsuccessful. To increase the probability of a successful installation, sediment movement through the reach should be ascertained (Section 10.3.1) and areas of localised scour and deposition identified (Section 10.3.2) through appropriate hydraulic and sediment-transport analysis.

10.3.1 Sediment transport

Understanding the dominant geomorphic processes operating in different sections of a river's catchment is a critical component of instream habitat design (Brierley 1999). There is no universally accepted approach that can be applied across catchments, because different reaches are subject to different biogeomorphic processes (Erksine & Webb 1999). Each reach needs site-specific analysis (Chapter 7) by a trained geomorphologist (Kondolf 1995a), in the absence of which a geomorphological classification system may be used. Kondolf (1995a) summarises some of the classification systems used to place a reach into the context of its surrounding catchment. If the catchment factors controlling the present status of a reach are not understood, attempts to manipulate instream habitats or fluvial processes may disregard forces that could undermine the intended goal. It would not make sense, for example, to introduce an artificial riffle (Section 10.5.1) into a sand-bed river, where there is a large likelihood of the riffle soon becoming covered with fine sediments.

Sediment supply, transport and deposition are three such catchment factors, and should be considered when undertaking a geomorphological reach assessment (Ferguson 1999). Evaluation of the movement of sediment through a reach is fundamental to the installation of rehabilitation structures since the amount and type of sediment supplied to a reach, which is itself a balance between transport energy and sediment load, has an influence on channel stability (Fogg & Wells 1998) as well as the stability of the instream structure. Generally a lack of available sediment relative to stream energy results in erosion of sediment from the channel. Conversely, an oversupply of sediment relative to stream energy usually results in deposition. If the installation of an instream structure is likely to alter the cross-sectional shape of the channel and this present shape is the desired one, the reach should be analysed to ensure that the upstream sediment loads being transported through it are not materially affected. This can reduce the likelihood of erosion and/or deposition taking place, thus helping to maintain the stability of the installed structure and the channel.

A sediment budget that accounts for sources and sinks of sediments can be calculated to help identify areas that are likely to be subject to erosion or deposition. Its purpose is to quantify processes of erosion, deposition, and transport within the catchment so that the amount of sediment delivered to the river, transported via the channel network, and deposited in lakes, reservoirs and wetlands can be estimated. Fogg & Wells (1998) reviewed the literature on sediment discharge formulae.

10.3.2 Scour and deposition

Identifying areas susceptible to scour helps to gauge the effectiveness of a proposed structure and also might highlight potential undesirable consequences. The formation of scour holes around an object may be related to the average velocity of the water, which is equal to the discharge divided by the cross-sectional area (Rutherfurd *et al.* 2000). For any one discharge, as the cross-section area decreases, the average velocity will increase. Similarly increases in the cross-section area will lead to a reduction in average velocity. Generally, an increase in velocity at one point in a channel will lead to a corresponding decrease at some other point, and thus further points that could be affected should be taken into consideration at the design stage.

Objects that have little impact on the cross-section area, such as a single boulder in a large river, will have little overall effect on scour and deposition patterns. Locally, however this may result in scour holes forming downstream of the object, due to the formation of high-velocity eddies around the structure, and depositional bars forming downstream of this scour hole. Rutherfurd *et al*. (2000) provided hydraulic guidelines for predicting the location of scour holes around obstructions.

Obstructions have other effects as well as causing downstream scour pools. If the obstruction covers the complete width of the channel, such as would happen with a weir or log dam, a backwater pool will form upstream of it (Rutherfurd *et al*. 2000). The depth of the pool will be a function of the height of the structure, the composition of the substratum and the ability of the flow to transport sediment. Rivers transporting large quantities of sediment will be susceptible to in-filling of backwater pools during low flow conditions, due to a reduced ability of the flow to transport sediment through the pool.

10.4 Rehabilitation of rivers with dams

Instream structures such as dams alter the flow of water, sediment, organic matter and nutrients, thereby affecting the physical and biological nature of the downstream river ecosystem and its riparian and floodplain zones (Fogg & Wells 1998). Upstream dams have been shown to reduce the supply of fine sediments such as gravel in Californian rivers (Kondolf 1998). Rehabilitation of these rivers involved either dumping gravel downstream of the dam to be redistributed futher downstream, or using the gravel to construct riffles below the dam. Those riffles that were constructed without consideration of the geomorphology of the site, the surrounding catchment influences and the sediment transport regime downstream of dams, were washed away with low return periods of 1.5 years.

River corridors can be rehabilitated to varying degrees through modification of the management and operational procedures of dams, or through complete removal of the dam. A study on the environmental impacts of different dam removal options for Rodam Dam on the Ocklawaha River in north-central Florida gave four options (Shuman 1995).

- *Full restoration of the river downstream of the dam*. Removal of the dam and all its associated structures, full restoration of the natural flow regime and of the inundation and sediment dynamics of the floodplain, and return of the downstream channel to its pre-construction conditions.
- *Partial restoration (rehabilitation).* Partial rehabilitation of the flow regime, floodplain and downstream channel through limited removal of structures.
- *Partial retention*. Reduction of the size of the reservoir and rehabilitation of that portion of the impounded section thus exposed.
- *Full retention*. Retention of the dam and reservoir, with active management of fish and wildlife resources.

The most important concerns related to the removal of Rodam Dam were sediment transport and floodplain dynamics (Shuman 1995). Although sediment transport invariably occurs after the removal of a dam, the amount of sediment released during and after dam removal is determined by the volume of sediment stored by the dam and the dam removal procedure. Although Shuman (1995) described many of the issues surrounding the removal of Rodam Dam, no decision has yet been made regarding which of the above four options will be applied.

Partial restoration of river corridors downstream of dams can be achieved by release procedures that mimic the natural hydrography, or desirable aspects of it (Fogg & Wells 1998). Water released from a dam to maintain a desired level of ecological functioning of the downstream reaches may be termed an instream flow (Kondolf 1998) or environmental flow (King *et al.* 2000). Historically, most attention focussed on a so-called minimum flow for maintaining the downstream ecosystem, but increasingly, high-flow releases or flushing flows are seen as equally important, because of their roles in channel maintenance and fish migrations and spawning.

Hill *et al*. (1998) described rehabilitation of the downstream reaches of the Owens River Gorge, California, using flushing flows. These reaches, downstream of an hydroelectric power plant, have been devoid of riparian vegetation and fish since the inception of the power station. The ten-year project aimed to increase complexity of the downstream riparian and aquatic habitat by using incremental increases in pulse (flushing) and base flows. The base flows provided the basic physical stability for the ecosystem whilst the pulse flows acted as ‘natural’ disturbances. The initial pulses were small enough to prevent erosion of landforms but large enough to facilitate downstream movement of sediments, re-create scour pools and incise banks. After each pulse, recovery of the landform vegetation was recorded, as was the response of physical conditions in terms of microhabitat development. The different sized releases allowed the banks of the downstream reaches to stabilise and vegetation to establish and mature. Larger and larger floods were released as the downstream reaches became more stable.

Many other examples now exist of environmental flows being used to aid river rehabilitation. Examples can be found in the World Bank *Good Practice Briefs* on environmental flows (World Bank in press), in the SADC report on *Environmental Sustainability in Water Resources Management* (Hirji *et al*. 2002), in various thematic reviews produced for the World Commission on Dams (WCD 2000) and on an array of Internet sites.

10.5 Instream rehabilitation structures

Most attempts at rehabilitating instream habitat are based on providing a stable channel and then increasing the hydraulic diversity (Rutherfurd *et al.* 2000). Hydraulic diversity can be increased, for instance, by creating alternating scour pools and high-velocity riffles. Other methods of instream rehabilitation include overcoming barriers to anadromous fish (saltwater fish that migrate up freshwater streams to their breeding grounds) and re-instating spawning grounds for these and other fish species. There are many options available to the stream manager, some tried and tested and others purely experimental. These fall within three main categories: structures that traverse the channel (Section 10.4.1); structures that partially intrude into the channel (Section 10.4.2); and structures that overcome barriers to fish migration (Section 10.4.3). These are outlined below, and some are also dealt with in Chapter 8.

10.5.1 Cross-channel structures

As noted in Section 10.3.2, structures that run across the entire length of the channel create an upstream backwater pool and a scour pool and deposition bar immediately downstream. Such structures are most often used to stabilise the bed in incised streams and to prevent or halt the upstream migration of nick points (head cuts) (Rutherfurd *et al*. 2000). Examples include weirs or sills (plunges), log dams, and artificial riffles. The most successful structures are those that tolerate high shear stress conditions. Rutherfurd *et al*. (2000) highlighted some guidelines for full-width structures:

- they should be constructed in straight sections of the channel so the distribution of velocities across the channel is uniform;
- the ends of the structure should be built into a stable bank, which can often be identified by the presence of well established vegetation;
- the bed should be stabilised both upstream and downstream of the structure, perhaps by using rock rip-rap;
- several smaller structures should be constructed rather than one large one, as the latter may present a barrier to fish movement;
- spacing between successive structures should be the same as that of natural riffles, which is often given as 5-7 times the channel width;
- well-graded quarry rock should be used to minimise infiltration of water into the bed of the stream, with larger material placed on top to create physical and hydraulic diversity.

Weirs, log-dams and sills

Weirs, log dams and sills dam water and stabilise the bed of the channel, thus providing a control point against upstream migration of nick points. Weirs built for this rehabilitative purpose commonly consist of large boulders placed across the channel, with smaller rocks packed into the gaps (Rutherfurd *et al*. 2000), or a wall of gabions. Use of gabions allows the wall to be shaped exactly as wished, as well as providing attachment points for vegetation. Gabion mattresses are time-consuming to build, but can be created without heavy machinery.

Log dams of a single log or stack of logs are a possible alternative to rock weirs. Their life span is surprisingly long if hardwoods are used. Completely submerged wood has a longer lifespan than wood that is alternately exposed and wet (Rutherfurd *et al*. 2000).

Logs placed to meet in the centre of the channel at a v-shaped angle form a sill (Rutherfurd *et al*. 2000) that steers low flows into the centre of the channel. Sills and log dams are cheaper than rock weirs and have a more natural appearance, but are more prone to failure by being undermined since they have a steeper face than rock structures. Generally full width rock structures are wider (longitudinally down the length of the river) than are log dams or sills, which tend to consist of between one and five logs arranged in a pyramid. The slopes of upstream and downstream faces of cross-channel rock structures are thus lower than those of the narrower sills and log dams. Rock structures are also coarser in texture than are the smoother log structures, with consequent higher surface areas and greater energy dissipation down the face of the structure. Even steep-faced rock structures dissipate energy as the water moves down the face (Rutherfurd *et al*. 2000). Designs need to accommodate this shortcoming of sills and log dams, by including plans to stabilise the downstream section against undercutting.

Artificial riffles

Artificial riffles are rock structures designed to replicate the shape of natural riffles. They are used to produce upstream runs or pools and provide fish spawning habitat. They may be designed specifically to allow free passage for migrating fish (Rutherfurd *et al*. 2000), by minimising the amount of water moving through the interstitial spaces between the rocks of the riffle and maximising that moving over its surface. This is achieved by using angular rock of differing sizes rather than of a uniform size, which increases the packing efficiency and reduces the number of interstitial spaces. Artificial riffles may also be used in much the same way as weirs to stabilise the riverbed, when they are usually designed as permanent structures fixed in one place. Because they have to be able to resist erosion, they are usually built of larger rocks than occur in natural riffles. Some artificial riffles mimic natural ones: they are designed to be mobile and thus require the continued addition of bed material as the riffle moves along the channel (Rutherfurd *et al*. 2000). Riffles may be created naturally if an artificial source of substrata with particles the same size as the naturally occurring material is deposited at intervals approximating the natural spacing of riffles. Other guidelines when building artificial riffles are as follows.

- Newbury and Gaboury (1993 cited by Rutherfurd *et al*. 2000) advise that the downstream face of the riffle should have a gradient of less than 1:20. This particular gradient has been used successfully in North America to allow free passage for migrating salmonids and trout.
- Do not build the riffle too high, since water will tend to pass through the structure rather than over it, again forming a barrier to fish passage (Rutherfurd *et al*. 2000).
- Build consecutive riffles between three and ten channel widths apart, at an average of six channel widths (Fogg & Wells 1998).
- Construct riffles in reaches where they would normally occur i.e. where they stand a better chance of long-term survival.

Artificial riffles are often used to increase the complexity of riverine habitats by, for instance, introducing fish spawning areas. These together with other instream structures such as groynes and deflectors (Section 10.5.2), can be used to provide fish with a variety of complex habitats (Schiemer & Waidbacher 1992). Re-instating riffles or adding artificial riffles has also been reported as effective in increasing biodiversity in degraded rivers, yet little study of the consequences of such interventions has been made. One such study compared the taxa found in natural riffles with those in artificially constructed riffles in the channelised Harper’s Brook, a tributary of the River Nene, in the UK. Ebrahimnezhad & Harper (1997) showed that the main taxa in artificial riffles were characteristic of natural riffles, and concluded that properly constructed artificial riffles improve the habitats and increase the biodiversity of macro-invertebrates to levels similar to those of natural riffles.

10.5.2 Partially intruding structures

Structures that partially obstruct currents are used to change the direction of flow. They may be used to protect unstable banks, create localised scour or deposition, re-create a meandering sinuous profile or create low-velocity zones that help to promote the establishment of vegetation (Rutherfurd *et al*. 2000). The structures may be impermeable (groynes, deflectors and submerged vanes) or permeable (retards and jacks).

Groynes

Groynes are used to manage erosion. They are built to abut perpendicularly into the flow, upstream of an eroding bend. Their main function is to direct flow away from an unstable bank, thereby reducing the velocity and shear stress in the vicinity of the point of erosion.

Deflectors

Deflectors are in-stream structures that change the direction of flow. They are usually low structures with minimal effect at bankfull discharge, and so provide minimal bank protection. Their main purpose is to create habitat by scouring out pools and depositing bars. They are ineffective in low-energy systems. The numbers and positions of the pools and bars depend on the placement of the deflectors: they may be angled upstream or downstream and used single or in multiples to produce a variety of hydraulic effects. Rutherford *et al*. (1999) provide a good summary of the range of possible conditions created from different orientations of deflectors.

Deflectors placed 5-7 channel widths apart will aid creation of a naturally sinuous channel (Nelson *et al*. 1978 cited by Rutherfurd *et al*. 2000).

Submerged vanes

Submerged vanes are deflectors that are not connected to the stream bank but act in the same way, causing pools to scour and sediment bars to form, the position of which depends on the placement and number of vanes used (Rutherfurd *et al*. 2000). They are designed to modify the structure of the riverbed and increase physical heterogeneity by creating these localised pockets of scour and deposition. Submerged vanes are recommended for low energy systems (Hey 1994 cited by Rutherfurd *et al*. 2000) such as lowland sand and gravel systems where they can effectively redistribute sediment. Single or double vanes create localised variations in scour and deposition patterns, while fields of vanes in parallel formations can initiate larger-scale channel bed changes. Submerged vanes are potentially powerful tools but are complex hydraulic structures requiring detailed design (Rutherfurd *et al*. 2000).

Retards

Retards are also channel-alignment tools used in erosion control, but differ from groynes by being permeable and so less efficient at diverting flow (Rutherfurd *et al*. 2000). Their most common function is to narrow and stabilise over-widened streams. Artificial benches placed in areas requiring stabilisation have poles sunk in that extend into the channel (Rutherfurd *et al*. 2000). Vegetation, mesh or logs attached between adjacent retards creates a permeable barrier to the flow. The retards aid permanent stabilisation of the artificial bench by encouraging deposition of sediment and subsequent establishment of vegetation.

Jacks

A jack consists of three concrete, wood or metal lengths joined at their midpoints, so that each length sits perpendicularly to the other two (Rutherfurd *et al*. 2000). They differ from retards by not being attached directly to the substratum, but are anchored to the bank by a cable. Jacks are more porous than retards, but also serve to introduce roughness elements to the flow. They are useful in wide or shallow streams subject to high-velocity flow where scour, highly mobile sandy beds or incising streams is a problem.

10.5.3 Bypassing barriers to migration

Instream structures such as dams, weirs or culverts can present barriers to the movement of fish (White & O'Brien 1999). Culverts represent barriers because current speeds through them during high flows can be too high for fish to cope with, and shallow waters in low flows may provide insufficient depth. Culvert outlets at a higher level than the bed of the river can provide additional problems during low flows (Lewis *et al*. 1999). More modern instream structures allow for the free passage of fish and other aquatic biota. It may be possible to overcome barriers caused by existing structures by installation of a fishway. There are two main types of fishway. The first type are fully engineered fishways (White & O'Brien 1999; Larinier 2001) and are used to bypass large barriers such as major dams. These are major features that should be designed from the outset for such large structures, and are not considered further here.

The second types are rock-ramp fishways. These are suitable for most smaller structures, such as small dams, weirs and culverts (Lewis *et al*. 1999). Rock-ramp fishways may either modify the existing river profile, cutting through the weir, or may be built offstream to bypass the weir. Rock-ramp fishways resemble permanent artificial riffles (Section 10.5.1) and are constructed from graded angular rock (Lewis *et al.* 1999). Fishways are designed to allow fish passage up to 95% of the time and hence must have water of sufficient depth flowing over them for this period (Rutherfurd *et al.* 2000). For rock-ramp fishways, Newbury & Gaboury (1993 cited by Rutherfurd *et al*. 2000) suggest the use of a v-shaped notch to concentrate low flows. Lewis *et al*. (1999) provide the following guidelines for their construction:

- rock-ramp fishways should have a consistent gradient;
- they should have resting areas at 1m intervals of elevation;
- the fishway can be lined with a geotextile to maintain water depth and stabilise the bed below the ramp;
- fish should be able to exit the ramp upstream into quiet water;
- sediment movement through the fishway and downstream of the fishway should be monitored regularly and accumulating sediments removed.

One way of improving the fish-passage potential of a culvert is to flood its outlet. This can be achieved by constructing a downstream weir or riffle, which will create a downstream pool that floods into the culvert. Care should be taken that the new structure does not itself form a new barrier. Introducing elements of roughness into the culvert by attaching rocks to its floor provides areas of refuge and a range of velocities at different depths. The increased hydraulic diversity inside the culvert will allow greater opportunities for fish movement at a range of discharges by providing low-velocity resting areas (Rutherfurd *et al.* 2000).

10.6 Summary

The goal of intervening within the boundaries of the channel is to create hydraulic, depth and substratum diversity by introducing structures that create habitat diversity and velocity variation. Rehabilitation of instream habitat will only work if the limiting factor affecting recovery at the site is habitat simplification. Prediction of the performance of a restoration technique at a particular site should be based on an understanding of its geomorphological character, the catchment influences, and the future stream power at the site. The most widely researched factors affecting the performance of instream structures are the movement of sediments, the scouring potential of the water, and the flow regime. Since instream structures rely on water movement to facilitate change, they need to be immersed during periods of low flow but also should not interfere with water movement at bankfull discharge. Localised areas of scour and deposition should be taken into account before a structure is installed, and a hydraulic analysis completed to estimate the velocities or shear stresses likely to be experienced by the structure. Three categories of instream structures used to manipulate and facilitate changes to aquatic habitats were discussed: cross-channel structures, partially intruding structures and fishways.

11. MONITORING AND POST-PROJECT EVALUATION

11.1 Introduction and definitions

Rehabilitation of river ecosystems is usually seen as providing substantial benefit to the environment and the surrounding community (Kondolf 1995). It could be argued that once rehabilitated, the condition of a particular reach will have been improved, and the situation will be better in some way than it was before. Questioning whether or not that is so appears to be a rare occurrence. This may be partly due to the paucity of 'products' against which to measure success; rehabilitating a river into a self-maintaining dynamic system does not easily produce something obvious to measure (Holl & Cairns 2002). Perhaps partly because of this, there have been few post-project evaluations of aquatic rehabilitation projects. Reasons given for the lack of post-project evaluation and monitoring (after Kondolf 1995; Kondolf & Michelli 1995; Lake 2001) include:

- regional ecological variation, because techniques easily applied in one river may not be applicable to another - each project may require individual assessment to suit site-specific needs;
- poor design and lack of advanced planning for many rehabilitation projects;
- post-project evaluation and the required techniques for it only being considered after the project is designed and implemented;
- preference among sponsoring agencies to fund tangible construction projects rather than intangible monitoring, with the latter being considered as research or experimentation and so less eligible for funding;
- the dynamic nature, and complexity, of river ecosystem functioning providing inherent difficulties in measuring rehabilitation success;
- a natural tendancy for a plan to lose momentum, to the detriment of later activities such as monitoring (Fogg & Wells 1998).

The lack of post-project evaluation is seen by many authors (Kondolf 1995; Kondolf & Micheli 1995; Fogg & Wells 1998; Lake 2001) as detrimental to the development of 'restoration ecology' as a science. Without such evaluation and dissemination of the results, rehabilitation lessons cannot be learned. A project should not be seen as complete until the condition and functioning of the rehabilitated river are assessed and adjustments made if necessary. The time frame for doing this could be months to years (Kondolf & Micheli 1995).

Post-project evaluation and monitoring differ. Monitoring may be defined as "…a continuous or periodic review of the activities within an identified project." (Watts & Fargher 1999). This implies a synthesis of information in a structured approach that includes three phases: planning and design, data collection and information management. Monitoring is usually used to assess the health of a river in a general way, most often through measurements of water quality or of pre-determined indicators of river health such as macro-invertebrates, fish and riparian vegetation. Recently there has been awareness that geomorphological and hydrological measurements should also be included in assessments of river health, as the river channel and flow regime largely dictate the formation and status of aquatic habitats utilized by all aquatic organisms.

Evaluation is a more recent concept that emerged in the 1950s, and may be seen as a subset of a larger monitoring programme. Evaluation is defined by Watts & Fargher (1999) as "…a process for objectively determining the appropriateness, efficiency and effectiveness of activities in the context of desired objectives." For example, the success of riparian vegetation in reducing erosion along a particular river reach could be evaluated. Evaluation thus compares collected data to pre-determined goals in terms of one or more performance indicators.

Even though many of the techniques applied in river monitoring are also applicable when evaluating river rehabilitation projects, the focus of evaluation is to determine whether the objectives of the rehabilitation project have been achieved. When used together, post-project evaluation and long-term monitoring are effective tools in natural-resource management, enabling the efficient design and implementation of projects (Watts & Fargher 1999).

Ecological monitoring is a science supported by an extensive literature. It can, and has been, the topic of entire books (Holl & Cairns 2002). It is thus not addressed here nor are monitoring and evaluation techniques, as these could be river and project specific. Instead, post-project evaluation aspects common to all rehabilitation projects are outlined below.

11.2 Essential steps in post-project evaluation

There has been no attempt to date to produce an all encompassing post-project evaluation methodology. The need for this, and suggestions for planning and evaluation, have been discussed by Kondolf (1995), Kondolf & Micheli (1995), and Holl & Cairns (2002). Steps in such a methodology have been synthesized from suggestions made by these authors.

11.2.1 Step 1: Secure resources within the budget

Generally, a rehabilitation project does not include a budget for post-project evaluation. When it does, the funds are often absorbed into some earlier phase of the project. Legal financial restrictions should be considered to ensure that evaluation is done within all rehabilitation projects.

11.2.2 Step 2: Define clear objectives that aid the evaluation

Reviews of rehabilitation projects have revealed a common lack of clearly stated goals (Holl & Cairns 2002). For example, the qualitative goal of "…restoring a biologically viable system…" is difficult if not impossible to evaluate. A more specific goal of, for instance achieving a certain percentage cover of riparian vegetation, can be more clearly and un-controversially evaluated.

All stated objectives should be compatible. As an example, the goal of re-instating variable floodplain inundation for development of a floodplain forest may not be compatible with maintaining a defined channel with mechanisms for flood control.

11.2.3 Step 3: Collect and synthesize baseline data from many sources

Reference data are usually used to design the target condition of the project reach, and are then used to evaluate success of the project in achieving this condition. Reference data may be historic, or baseline data collected before a site was damaged, or data from nearby intact and similar reaches. Often a combination of these data is used. Historical conditions should guide the design of the target condition as much as possible since they provide a temporal context in which to interpret the results of the evaluation. This will help minimize the confusion between natural or long-term or cyclical changes, and those resulting from project manipulations. For example, a channel cannot be assumed to be stable simply because no channel changes occurred during the construction phase of a rehabilitation project.

Where the condition of the river before human disturbance is not known, and there are no nearby reaches in good condition, a range of reference sites may be employed. These could illustrate a variety of successional stages across sufficient spatial scales to allow natural variability within the river to be understood.

11.2.4 Step 4: Design a study to help demonstrate and develop principles of rehabilitation

Each rehabilitation project should be seen as an opportunity to observe the effect of manipulating the river ecosystem. The development of principles is discussed in Chapter 15. Briefly, each project should be treated as an experiment, which includes control reaches with the same basic nature as the project reach. Conditions in the rehabilitated and control reaches could then be compared, with the same variables measured at the same time intervals, to distinguish between natural and man-made changes.

The design of the evaluation should reflect the overall objectives. For example, if the overall objective is to increase the cover of riparian vegetation, this should be translated in the design to a goal of achieving specific percentages of cover or species densities, and measurements taken accordingly. Given the experimental nature of rehabilitation projects and the dynamics of aquatic environments, such defined goals may have to be stated as ranges of acceptable variation rather than a single value. This can be challenging, as replicate measurements within one community may well be as little as 50% similar (Gauch 1982 cited by Holl & Cairns 2002). Setting the range of acceptable change too tightly might cause an undesirable change to be detected where none has occurred. On the other hand, setting the range too wide may result in important changes remaining undetected.

11.2.5 Step 5: Be aware of temporal and spatial scales

Temporally, for the purposes of post-project evaluation, monitoring may well have to continue for a decade or more after the rehabilitation. Sampling intervals can increase with between sampling occasions increasing with time. Most post-project monitoring, if funded at all, may well be limited to the first year or two after project completion. This is insufficient time for a rehabilitated river to manifest most changes: large channel changes may only take place after one or a series of high flows; riparian vegetation may take years to develop new communities; fish populations are subject to natural fluctuations and expected changes may lag because time is needed for habitat improvements to lead to increased abundances of aquatic invertebrates and other food sources.

Spatially, when designing an evaluation, a trade-off may have to occur between the frequency of sampling, the number of variables measured and the number of locations (Michener & Houhoulis 1997 cited by Holl & Cairns 2002). To the extent that budgets allow, the sampling locations should be distributed across the entire area for which inferences need to be made and not just in areas of easy access.

11.2.6 Step 6: Document all steps of the rehabilitation project thoroughly

Records of all decisions, together with their context and motivation, should be recorded, from the initial planning process to the post-project evaluation. The data at all stages from baseline studies to post-project evaluation should be collected using the same techniques. The evaluation protocol and location of monitoring sites should be documented to facilitate repeat measurements over the years. This documentation will facilitate smooth transitions when project personnel change or the overall plan needs revision. In addition, project documentation should integrate evaluation criteria into each phase of the rehabilitation plan. For example, evaluation of the appropriateness and stability of designed project structures may indicate a likelihood of failure well before completion of the project. In such a case, the structure could be adapted to before the project is completed. If, for instance, the substratum size of a constructed riffle is deemed to be too small, either as a spawning substrate or to prevent the riffle being washed downstream during a high flow event, larger substrata could be substituted.

11.2.7 Step 7: Acknowledge failures as lessons learnt

If relevant variables are measured during monitoring, using standardized techniques that are reproducible, the results will be valuable. Even if the rehabilitation project is eventually seen as a failure, its objectives and outcome will make a contribution to the science of rehabilitation. Others may learn from the mistakes made, and documents will be available for study to aid the development of rehabilitation concepts and principles. All projects can be seen as experiments from which the science of rehabilitation can grow.

11.3 Summary

There have been few post-project evaluations of aquatic rehabilitation projects. Those documented reveal a high percentage of failures. Lack of post-project evaluation is detrimental to the development of 'restoration ecology' as a science. By continual evaluation during and after a project, problems may be identified before they become prohibitively complex or expensive to correct. Without such evaluation and dissemination of the results, lessons cannot be learned that can contribute to rehabilitation science. A project should not be seen as being complete until the condition and functioning of the modified river ecosystem are assessed and adjustments made if necessary. The time frame for doing this can vary from months to years. When used together, post-project evaluation and long term monitoring are effective tools in natural resource management enabling efficient design and implementation of projects.

12 THE LOURENS RIVER

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Due to the large amount of data generated during the study period only a portion of it can be displayed. In this report only results from Site 1 are presented, as most of the channel changes took place there. The bulk of the data for both sites will be dealt with in the M.Sc. thesis of Miss Lindie Smith (in prep.). The thesis is entitled: The relationship between channel discharge, hydraulic biotopes and channel morphology in the Lourens River, Western Cape, South Africa. Unpublished M.Sc. thesis, University of the Western Cape, Bellville.

12.1 Lourens River Catchment

The Lourens River is 20 km long, rising at an altitude of 1080 m within the Diepgat ravine, also known as Watervalkloof, of the Hottentots Holland Nature Reserve. The river flows in a south-westerly direction through Somerset West before entering the Indian Ocean via a small estuary located in the north-east corner of False Bay (Cliff & Grindley 1982; Tharme *et al*. 1997). It is a perennial river, with a catchment area of approximately 140 km^2 (Tharme *et al.* 1997). The river has no major tributaries but is supplemented by minor tributaries (Figure 12.1). The upper catchment is vegetated with Mountain Fynbos but the foothill and coastal parts of the catchment have been extensively changed. Land-use includes pine plantations, vineyards and other crops, a piggery, a nursery, a sawmill and a mixture of residential and recreational areas. These activities have lead to the invasion of alien plants along riverbanks, poor water quality, reduced aquatic and riparian biodiversity and stream bank erosion (Tharme *et al.* 1997). The latter factors have resulted in recently observed channel changes or adjustments in the Lourens River, such as bank scouring and undercutting, channel erosion and channel migration and the formation of bars and islands (pers. obs.).

Recent developments in Somerset West have resulted in large parts of the town now falling within the 50- and even 20-year flood lines. Urban developments such as the construction of houses, roads and parking lots limit infiltration after rainfall events, resulting in catchment hardening and increased runoff into the Lourens River. An increase in both the frequency and intensity of flooding resulted in the 20-year flood lines becoming 5-year flood lines. Properties adjacent to the river that in the past were seldomly flooded are now regularly flooded (Tharme *et al*. 1997). Following floods experienced in the early 1990s, the Municipality of Helderberg realized that stormwater management and the provision of measures against flooding were inevitable (Tharme *et al*. 1997; Crowther Campbell & Associates 2000). Phase 1 of the Lourens River flood alleviation project therefore started in January 2002, with channel widening and protection barriers within the urban area.

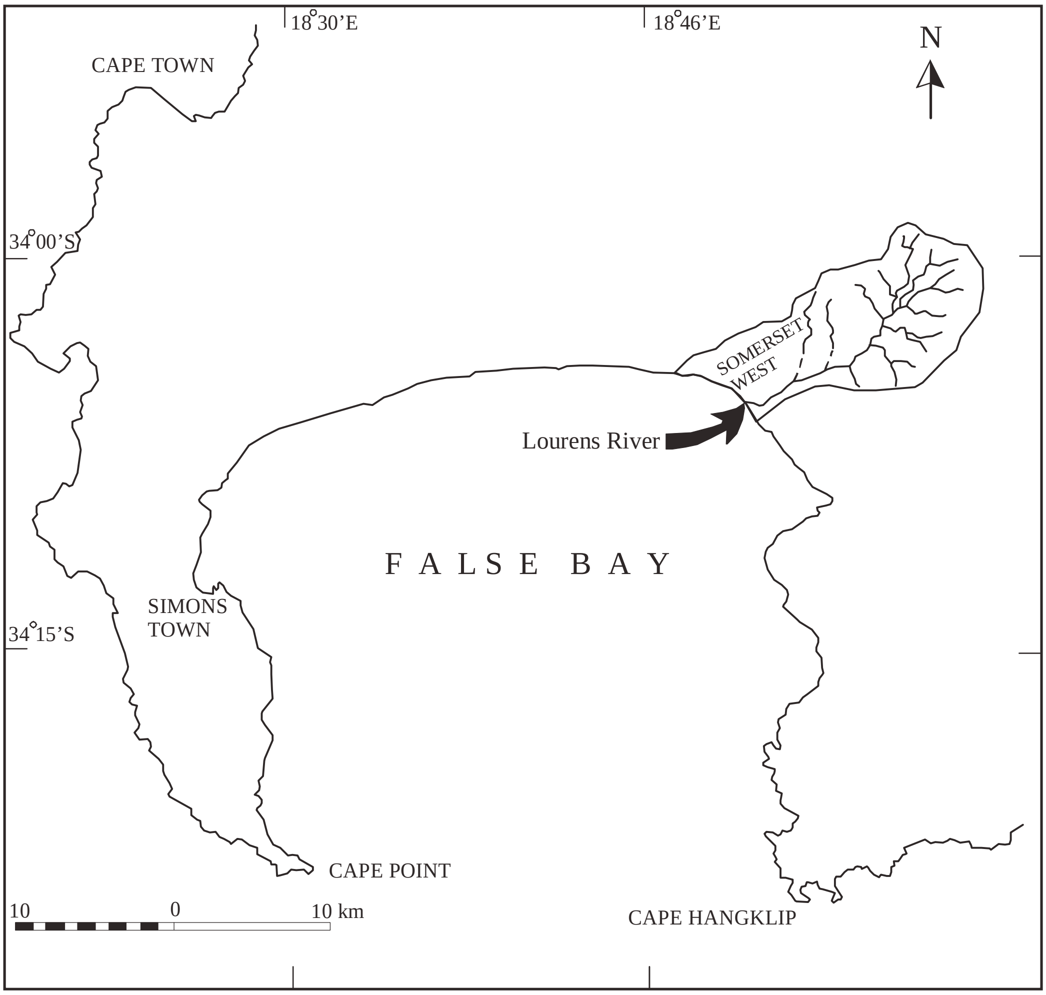


Figure 12.1 Location of the Lourens River catchment area (modified from Cliff & Grindley 1982).

12.2. Overall objective and work plan

The original objective of this project was to measure the response of the Lourens River to clearance of alien vegetation from its banks. When the planned clearance did not take place, an amended objective was to measure river response to flood mitigation measures within the urban area of Somerset West. When these also did not take place, the objective was again amended to make it independent of channelisation work. Instead, the project focused on the relationship between channel discharge, hydraulic biotopes and channel morphology. Rowntree & Wadeson (1999) have defined hydraulic biotopes (e.g. pools, runs and riffles) as spatially distinct instream flow environments with characteristic hydraulic and sedimentary attributes that occur in morphological units (Table 12.1). Morphological units are the basic structures recognised by fluvial geomorphologists as comprising the channel morphology and may be either erosional or depositional features (Rowntree & Wadeson 1999). Two sites in the lower Lourens River with different reach and site characteristics were used for a comparative study of channel morphology and local hydraulics. The main objectives of the study were to:

- determine the influence of substratum and discharge on the flow characteristics of 12 hydraulic biotopes (three runs and three riffles at each of two research sites);
- determine the direction of change of different kinds of hydraulic biotopes in response to changing discharge;
- determine the relationship between hydraulic biotopes and the distribution of plants and animals at the sites;
- examine the extent to which channel morphology is influenced by flow changes;
- assess vegetation composition and distribution and relate that to changes in channel morphology.

12.3 Study area and site selection

The river lies within the Winter Rainfall Region of the Western Cape and consequently most of the discharge occurs during the winter months (Cliff & Grindley 1982). Mean annual precipitation ranges from 1500 mm in the upper catchment to around 600 mm in the lower reaches of the catchment (Midgley *et al.* 1994). Very high rainfall was experienced in the area during the study in the winter of 2001 (Figure 12.2). Estimated mean summer temperatures lie in the range of 18 °C in the higher areas to 24 °C on the lower slopes. Corresponding mean monthly winter temperatures lie between 5 °C and 14 °C. Mean annual runoff (MAR) for the Lourens River is in the order of 122×10^6 m^3 of which 13% occurs in summer and 87% from April to October (Tharme *et al.* 1997).

The river rises in the deep kloofs of the Hottentots Holland Mountains, passes through shallow valleys surrounded by undulating hills and finally cuts across the flat coastal plain before entering the sea. The upper catchment is underlain by Table Mountain Group sandstones, the middle catchment by Pre-Cape granites and Malmesbury Group shales while Tertiary/Quartenary alluvial clays and aeolian sands dominate the foothill zone and coastal plain (Cliff & Grindley 1982; Tharme *et al*. 1997). In the upper reaches a mixture of fynbos and renosterveld dominate the natural terrestrial vegetation. However, much of the natural vegetation in the lower reaches is depleted and disturbed, and exotic garden species dominate. Dense infestations of exotic species such as grey poplar (*Populus x canescens*), black wattle (*Acacia mearnsii*), kikuyu grass (*Pennisetum clandestinum*) and other species occur. Invertebrates found include dragonflies, mayflies, snails and limpets, worms, beetles and crabs. Indigenous fish species recorded in the river include Cape kurper (*Sandelia capensis*) and Cape galaxias (*Galaxias zebratus).* Introduced species such as the mosquito fish (*Gambusia affinis*), rainbow trout (*Oncorhychus mykiss*) and banded tilapia (*Tilapia sparrmanii*) are also known to inhabit the river (Tharme *et al*. 1997). The study area locations are shown in Figure 12.3.

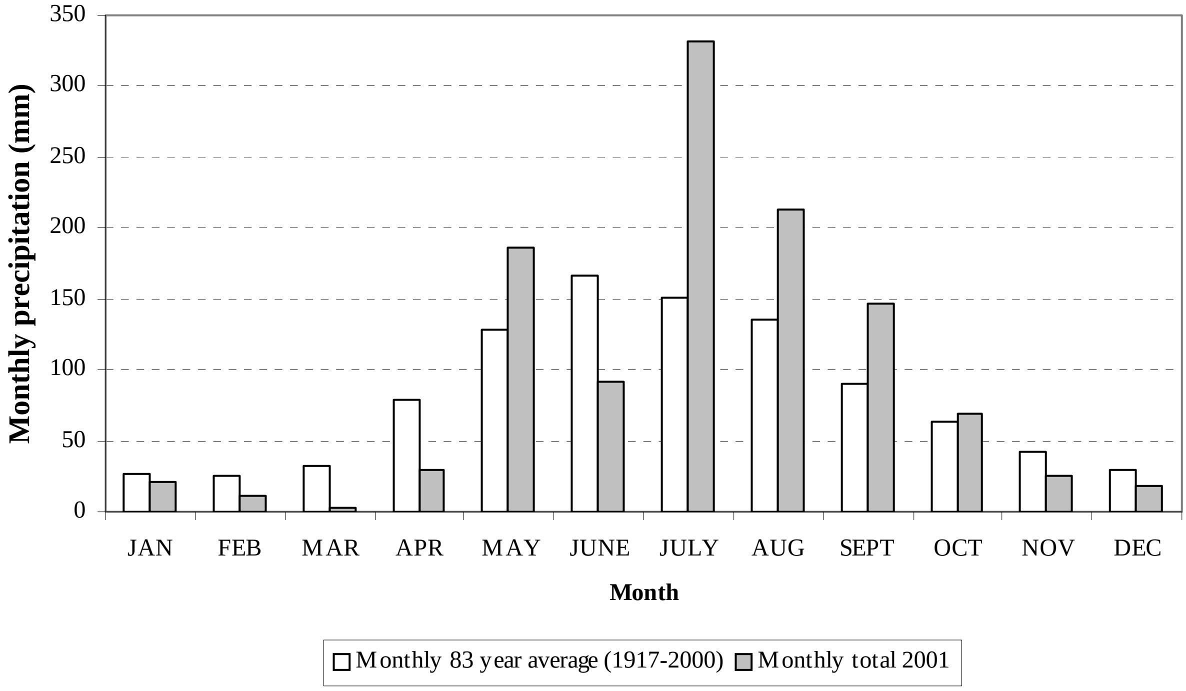


Figure 12.2 Precipitation data for the foothill zone (Lourensford Estate, Somerset West, unpublished data).

12.3.1 Site 1

This site in Radloff Park was selected on the basis of representivity of diverse geomorphological and physical characteristics, degree of disturbance, as well as accessibility. It was one of the few reaches along the Lourens River that lacked major channel modifications such as canalization and infilling, except for the first ten meters of the left bank which had gabion mattresses. In this reach, the river forms a boundary to Radloff Park, a recreational area that is located in the foothill zone and opposite Morgenster Estate. Site 1 had a lateral cobble/gravel bar and terraces on the right bank of the upper part of the reach, and short riffle/pool/run transitional hydraulic biotopes. These biotopes differ in their substratum composition, depth and flow velocities, at any one time, and vary dynamically with flow (Table 12.1). Riffles for example, are typically topographical high areas of channel beds, comprising coarse materials (cobbles and gravels) and have turbulent fast flow. In the case of runs however, there are no obvious changes in streambed gradient. Runs are characterized by medium rippled flow with no broken water on the surface, occurring on any substratum apart from silt (Rowntree & Wadeson 1999). They often form a transition zone between riffles and downstream pools. Pools are characterized by finer bed materials than riffles and are normally found at topographical low areas of the streambed (Rowntree & Wadeson 1999).

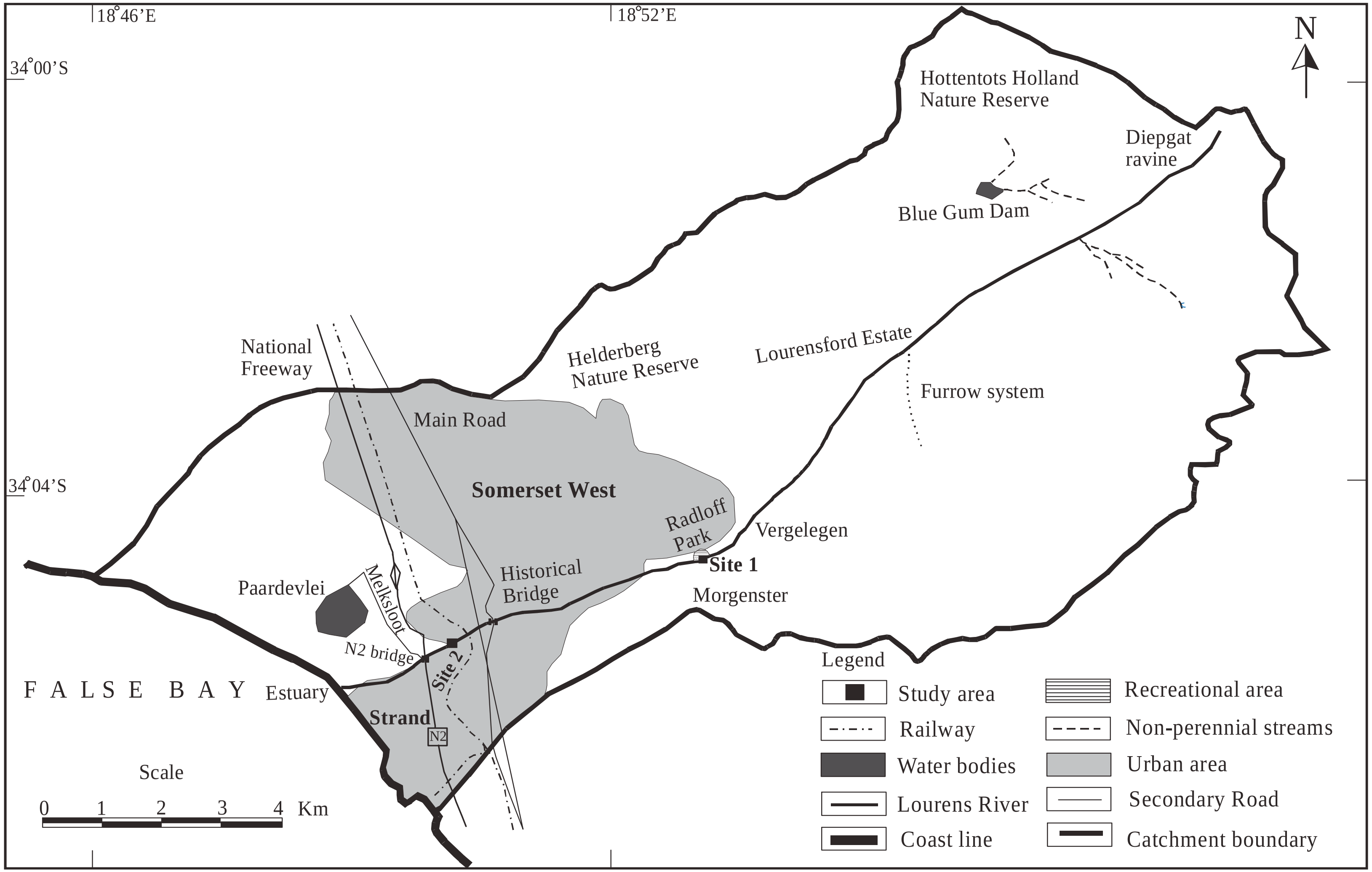


Figure 12.3 **The Lourens River catchment, showing the location of the two study sites in the urban area**.

Table 12.1 The definitions of hydraulic biotopes (after Rowntree 1996; Wadeson & Rowntree 1998).

Hydraulic Biotope	Definition
Riffle	Riffles may have undular standing waves or breaking standing waves and occur over coarse alluvial substrates from gravel to cobble.
Run	A run is characterised by a rippled or surging flow type and occur on any substrate apart from silt. Runs are often the transition between riffles and pools.
Pool	A pool is in direct hydraulic contact with upstream and downstream water but has barely perceptible flow.

The site was approximately 90 m long, with a single channel that was confined and gently meandering. The banks were steep, eroded and consisted of sand and cobbles. The wetted perimeter varied between 2-10 m along the length of the site in the dry season. Reach morphology could be classified as run-riffle. The riverbed substratum consisted mainly of small to medium-sized cobbles, with occasional large cobbles and a few small boulders. Sand patches occurred along the right bank. The tree canopy was largely closed, opening in a few areas only, and comprising primarily alien trees such as poplars and weeping willows. Marginal vegetation in the form of kikuyu grass, sedges and weeds were apparent at this site. Immediately upstream, gabion mattresses formed part of the right bank. Downstream from the site, the river was constrained by a concrete wall on the left bank, designed as a flood protection measure.

12.3.2 Site 2

This site was originally chosen because of a proposed flood alleviation project as well as its accessibility. It was situated in the upper coastal plain zone, upstream from the Sergeant Street Bridge to downstream of the N2 Bridge, in an urban suburb. The former floodplain of the river represented by this site had been built upon, resulting in numerous modifications to the natural river system. Parts of the right bank were lined with grassed gabions *Pennisetum clandestinum* (kikuyu grass), while other sections had unnaturally high banks with tall narrow-stepped gabions. The site had a single, mildly meandering channel that was constrained by bank-side developments. Shallow to moderate deep pools alternated with runs and riffles. The wetted perimeter varied between 4-9 m along the length of the site, during the dry season. The substratum comprised mainly large to small cobbles, with considerable amounts of sand and gravel. The tree canopy was largely open, with a few large weeping willows providing patches of shade. The river fringes consisted mainly of manicured gardens, grassed down to the banks. There was no instream vegetation at this site. The site was approximately 150 m downstream from Sergeant Street Bridge. The geomorphological zones

(Rowntree & Wadeson 1999) and study sites are indicated on the longitudinal profile of the river (Figure 12.4).

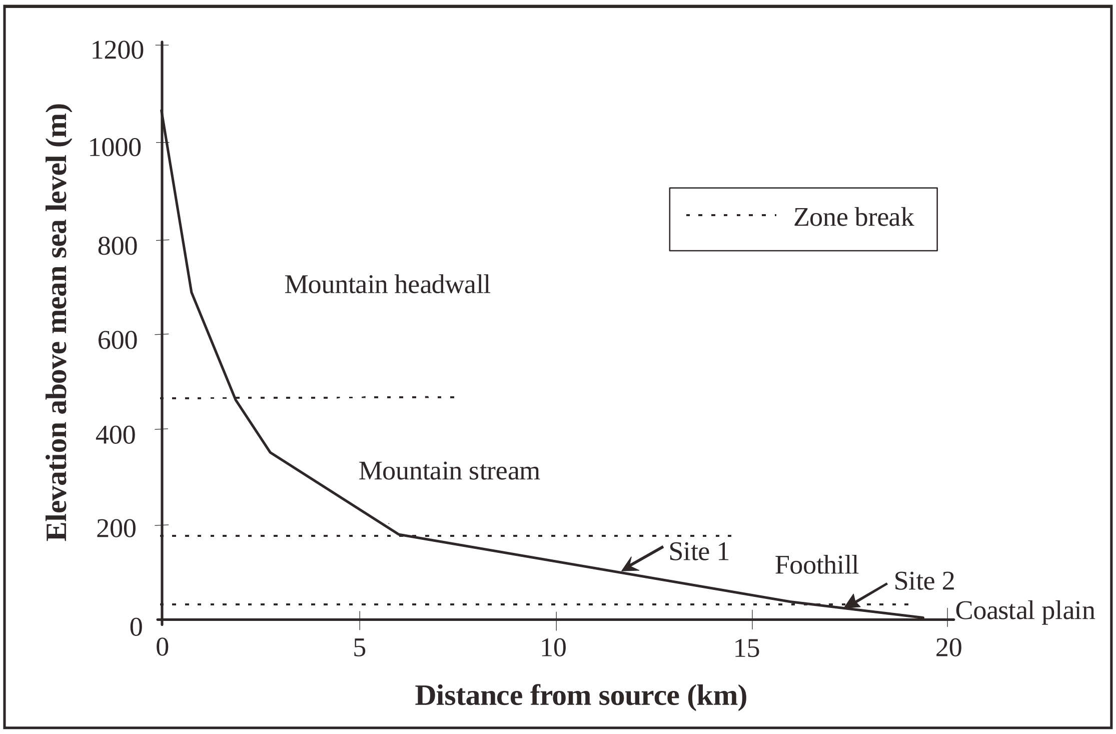


Figure 12.4 Longitudinal profile of the Lourens River indicating locations of the geomorphological zones.

12.4 Methodology

This section outlines the methods and the techniques used in generating the data sets for the different parameters. At each site six kinds of activities were completed. Firstly, cross-sectional data were obtained through surveying and were used to provide input into channel morphology changes. Secondly, habitat maps were drawn based on visual estimations of substrata and flow types. Thirdly, vegetation composition and distribution were assessed along several of the cross-sections and related to changes in channel morphology. Fourthly, aquatic invertebrates were collected within each hydraulic biotope and used to indicate the general health of the river. Fifthly, discharge readings were employed in an attempt to explain channel and hydraulic biotope changes. Lastly, sediment sizes were correlated to the different kinds of hydraulic biotopes. Data were collected at the two sites in a summer and winter sampling programme, starting at the beginning of 2001 and continuing to the end of the 2002 summer season. Table 12.2 outlines the dates when the different parameters were sampled, drawn or measured.

Table 12.2 Dates when the different parameters were sampled, drawn or measured for Site 1 and Site 2.

Parameter sampled, drawn or measured	Summer 2001		Winter 2001		Summer 2002	
Site	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
Habitat maps	April	January to February	July	July to August	February	January to February
Vegetation	April	February			February	January to February
Macro invertebrates	May	May			February	January
Cross-sections		January, June and November	June and August		March	March
Discharge	May	April to May	June, July and September	June, July and September		
Substratum composition			August	August		

12.4.1 Cross-sections and transects

Channel geometry and the nature and condition of the banks were measured through repeat surveys of cross-sections. Nine cross-sections were surveyed at Site 1 using the tachometry method of optical distance measurement. Cross-sections were chosen on the basis of representivity of mainly run and riffle hydraulic biotopes, to give an accurate representation of geomorphological features of the reach and to measure characteristic flow hydraulics at observed discharges. Pool hydraulic biotopes were avoided because of the difficulty of accessibility during peak discharges (cross-sections 1.6 and 1.9). Cross-sections represented three runs and three riffles. Multiple hydraulic biotope cross-sections were chosen because no two runs or riffles are identical, especially when different channel discharges are involved (Rowntree & Wadeson 1999). The same cross-sectional profiles (Transect 1.3, 1.7 and 1.9) were used as vegetation transects describing the terrestrial vegetation pattern. The positions of all the cross-sections were indicated on the site maps. Cross-sections were positioned to best capture any possible channel changes.

An electronic theodolite, Leica TC307 model, was used to obtain readings reflected off a prism (Standard Leica prism) fixed onto a staff. The horizontal distance to the staff, the difference in surface elevations and the location in reference to other points were automatically derived. This included the co-ordinates of all the points along a cross-section. Cross-sections were marked using fixed points in the form of metal stakes inserted into the ground. The fixed points were used for re-surveying. The survey system (points) of Site 2 was fixed to the National Survey Coordinate system. Site 1 was not linked to the system because it was too far away from town survey marks and its left bank consisted of loose sand that made stable positioning of the metal stakes impossible. This meant the site was not Geo-Referenced although it had its own local co-ordinate system (M.C. Briers, Survey Department, University of Cape Town, pers. comm., 2001 and 2002).

The theodolite was also used to map the channel outline, plan and contour maps when the sites were surveyed in 2001 and 2002. Computer Aided Drafting (CAD) programs Autocad and Allycad were used to generate the channel outlines and profiles.

12.4.2 Habitat mapping

Biotopes are homogenous environments that satisfy the habitat requirements of biotic communities (Rowntree & Wadeson 1998). Habitat maps were drawn for each site, based on visual estimations of proportions of streambed materials of various sizes and flow types. Substratum size categories and features mapped are demonstrated in Tables 12.3, 12.4 and 12.5. Categories of flow types are shown in Table 12.6.

Table 12.3 Categories of substratum, adapted from Rowntree & Wadeson 1999; King & Schael 2001.

Class name (Wentworth scale)	Size range mm (b-axis)	Phi units (ø)	Guide line
Clay (Cl)	0.00006 to 0.0039	14 to 9	Mud
Silt (S)	0.0039 to 0.0625	8 to 4	Mud
Fines (F)	0.0625 to 2	4 to -1	Sand
Gravel (G)	2 to 64	-1 to -6	Finger nail to length of small finger.
Cobble (C)	64 to 256	-6 to -9	Wrist to halfway along finger; Inside elbow to wrist.
Boulder (B)	>256	>-9	Armpit to wrists; ground to waist; >length of tall person.
Bedrock (Br)			Slabs of rock.

Table 12.4 Geomorphological units (after Van Niekerk *et al.* 1995; Rowntree & Wadeson 1999).

Geomorphological Unit	Description
Lateral bar	Accumulation of sediment attached to the side of the channel, may occur sequentially downstream as alternate bars.
Island	Large mid-channel sediment accumulation that is rarely inundated.

Table 12.5 Adapted key for features used in subtratum mapping.

Feature	Description
Mud Block (MB)	A combination of clay and silt material.
Smooth Gabion (SG)	Wire mesh baskets filled with rocks. These flexible ‘‘mattress” structures are normally overgrown with riparian vegetaion.
Building Rubble (BUR)	Material such as bricks or cement slaps.
In-channel Woody Debris (IWD)	Fallen trees, twigs, logs or branches.
Marginal Vegetation (MV)	Refers to the vegetation occupying the area along the stream fringes that might be partially submerged or may protrude from the water.
Vegetation (V)	Plants growing in the riparian zone.
Organic Litter (OL)	Allochthonous plant material such as leaves, barks, flowers and fruits that fall from the canopy cover or which are washed or blown from the surrounding terrestrial vegetation into the river.
Algae (A)	Epilithic algal growth, algae attached to the surfaces of substrata (e.g. stones).

The distributions and proportions of different substrata were mapped according to particle size, from the most dominant particle size to the least dominant one. This implied that, when for example sediment size proportions were visually estimated, and cobbles covered 85%, gravels 10% and fines 5% of the bed, the sediment category was classified as cobble, gravel and fines. The degree of sorting of the sediment was thus mapped. Sorting refers to the grain size uniformity of the sediment. If

Table 12.6 Flow types (after Rowntree & Wadeson 1999; King & Schael 2001).

Flow type	Definition
No flow (NF)	No water movement.
Barely perceptible flow (BPF)	Smooth surface flow; flow only perceptible through the movement of suspended matter.
Smooth flow (SMF)	Still smooth surface; streaming flow takes place throughout the water profile; turbulence can be seen the upward movement of fine suspended particles.
Meduim rippled flow (MRF)	The water surface has regular disturbances which form low transverse ripples across the direction of flow; the degree of disturbance may vary form faint ripples to strong ripples.
Fast turbulent flow (FTF)	Standing waves present which break at the crest (white water).

one particular particle size dominates, the sediment is well sorted. If a mix of particle sizes occurs, the sediment is poorly sorted. Either surveyed maps or a grid of tapes were used as template for the maps. By laying a tape measure in a straight line along the length of the channel, channel outlines were mapped. The width of the channel was measured at right angles to this tape by using a second tape (King & Schael 2001). With the aid of a scanner, the field-drawn habitat maps were then imported into Coral Draw V 8, a computer-based drawing program, and digitized on screen. Color codes were added to create the final substratum and flow type maps. The actual areas of the different categories of substratum or flow types were calculated using a planimeter. Features mapped were:

- sediments (bedrock, boulders, cobbles, gravel and fines);
- flow types (no flow, barely perceptible flow, smooth flow medium rippled flow and fast turbulent flow);
- organic detritus/litter;
- marginal vegetation (at water's edge);
- gabions (smooth or stepped);
- building rubble (e.g. cement slabs);
- algae covering the substrata.

Canopy cover (Table 12.7) over the stream was estimated in the summer of 2001 and 2002. Habitat maps were employed in identifying hydraulic biotope changes and macro invertebrate sampling. The maps were re-drawn at intervals throughout the project (Table 12.2), to indicate how the hydraulic biotopes changed with season and due to floods.

Table 12.7 Canopy cover categories (Dr. Jackie King, Freshwater Research Unit, University of Cape Town, pers. comm.).

Category	Description
Shading (SH), closed canopy cover	The extent of cover of riparian vegetation over the stream.
Sunlight (SN), open canopy cover	No shading.

12.4.3 Invertebrate sampling

At each site, invertebrate samples were collected from a range of hydraulic biotopes. Invertebrate sampling was done according to a simplified version of SASS 4, as an assessment of the degree of impairment of water quality and the general health of rivers. SASS 4 procedure involved time limitations for taxa sampling and identification, specific sampling area sizes, and a different abundance rating scale (Thirion *et al.* 1995). A 250-μm mesh net was used in a back and forward sweep motion to sample the marginal vegetation and backwater areas. The animals were scraped off cobbles and boulders with a small brush into the net. Sampling areas differed in size and depended on the availability of habitats. For each sample, the content of the net was tipped into a sorting tray. The leaves and twigs were then removed, and the animals identified with the help of a manual (Thirion *et al.* 1995). The location for each sample was indicated on the habitat maps (Figures 12.6 and 12.7). Areas sampled varied from 1.0-1.5 m long and 0.2-0.6 m wide. Abundances were estimated using the following rating scale: 1 = 1 organism, 2 = 2 to 6 organisms, 3 = 7 to 20 organisms, 4 = 21 to 100 organisms and 5 = >100 organisms (Jackie King, Freshwater Research Unit, University of Cape Town, pers. comm.). Samples were then placed in jars, which were labeled inside and outside.

Samples were first fixed with 4% formalin and taken to the lab, where the formalin was replaced with 70% ethanol. Further identification took place in the laboratory. Data analysis involved entering the data into Excel and importing it into Primer V 5.0 (Clarke & Warwick 1994;Clarke & Gorley 2001). Cluster- and ordination analyses through multi-dimensional scaling (MDS) were conducted in Primer V 5.0 for both the vegetation and invertebrate data. The resulting 2-dimensional MDS plots and dendrograms were used to group similar biological communities. The distance between samples reflected dissimilarities in community structure: samples that were close together had similar communities and those more distant had less similar ones. The macro invertebrate data were transformed using the presence/absence transformation. A transformation is used to weight the contributions of the different taxonomic groups and the choice of transformation is a biological not a statistical one. With no transformation, the two/thirds most common groups will dominate the similarity matrix. Presence/absence, which can be regarded as the most severe transformation possible, down weighs the contribution of the common species in relation to the rarer ones. In the presence/absence transformation 1 (presence) or 0 (absence) represents the data matrix and Bray-Curtis similarity is computed. All species have equal weight, whether rare or abundant (Clarke & Warwick 1994; Clarke & Gorley 2001). The calculated stress value provided a good means of assessing the reliability of the MDS ordination. According to Clarke & Warwick (1994) a stress value of <0.05 gives excellent representation with no prospect of misleading interpretation. A stress value of >0.3 could give a misleading picture and should therefore be treated with great caution.

12.4.4 Vegetation sampling

Vegetation data were collected along the whole length of selected transects. Cover and abundance values were estimated for species inhabiting $1m^2$ grids at each site, using the Braun-Blanquet cover-abundance scale methodology (Gordon *et al.* 1992). In each grid, the cover of each species was estimated as the proportion of the ground occupied by a perpendicular projection of the aerial parts of each species expressed as a percentage. The maximum height of each species in each grid was also estimated, together with the percentage of live individuals in each. Live plants had a high percentage value whereas dead or dying ones had a low percentage value. Samples of the same species were collected from the surrounding areas, pressed and identified by specialists. The vegetation data were first imported into a vegetation database program, Turbo Veg V.199b, exported to Excel and then imported into Primer V 5.0 for analysis. Turbo Veg V.199b was used to enter information such as cover percentages and plant names. Excel was used for table sorting and preparation for printing. As described in Section 12.4.3, Primer V 5.0. was used to group similar biological communities. Grids without vegetation such as grids in the channel and outliers were removed from the data set and excluded in the analysis process. Outliers were identified as species that were isolated from the rest of the data set and had a disproportionate effect on the clusters that were formed. They were removed to allow better examination of the remaining clusters.

12.4.5 Hydraulic characteristics

In order to understand the relationship between high flow events and channel changes, information on discharge was necessary. Since most of the channel and biotope changes seem to occur during flood events, discharge was only measured during the flood season in winter. No readings were taken in summer during base flow conditions. Field observations, accompanied by detailed notes, were made during periods when it was impossible to enter the river to take discharge readings. When flow levels dropped to a workable level, discharge readings were taken on six occasions during the winter of 2001. Discharge data for both sites were collected on the same day, during the same flow event. Water depth was measured using a top-setting wading rod, and average velocity with a Price AA or pygmy current meter (Gordon *et al.* 1992). At each discharge cross-section, a tape was stretched tight across the channel above the water surface and attached to either bank. Approximately ten readings of depth and average velocity were made along the tape at intervals ranging from 0.5m-1m at 60 % depth. Discharge calculations were summarized using an Excel spreadsheet. The velocity area method was used to calculate discharge (Gordon *et al*. 1992). Each measured discharge was used to calculate the average discharge and standard deviation for that day. Flood levels were recorded along selected cross-sections at both sites and a detailed photographic record was maintained.

12.4.6 Substratum composition

Initially, it was planned that bed material would be sampled before and after the winter of 2001 to determine the sediment sizes for the different hydraulic biotopes at each site, and relate this to the different flow types. Due to logistical problems of sampling the bed material, laboratory time necessary to process the samples and time constraints caused by the problems experienced at the beginning of the study (Section 12.2), it was decided to take samples of bed and bank materials on one occasion in August 2001 and to employ the results in a descriptive capacity only. While it is acknowledged that substratum characteristics can change over time, visual and photographic evidence presented little evidence to suggest major changes in the composition of bed material during the study period. As reference for sampling the bed material transect 1.3 with a variety of substrata was chosen. The Wolman Pebble Count technique, using a grid system (quadrants) along profile 1.3 was used for determining the size of the gravel fraction while sieve and settling tube analysis were used for the sand, silt and clay fractions. The following procedure was followed. A tape was stretched tight across the channel above the water surface and attached to either bank. A wire square grid ($1m^2$) was then placed next to the tape and used to define the sample area. A gravelometer (template) was used to measure the particle diameter of streambed materials in the medium gravel to boulder size range in five quadrants. A minimum of 100 particles was measured in each quadrant. Scoop or grab samples were collected from the banks and bed where particle sizes were in the sand to fine gravel range and placed into 500g jars. These were then taken back to the laboratory for dry sieving and settling tube analyses. The results were manually entered and processed in an Excel spreadsheet. Grain size cumulative curves were constructed to indicate the percentage bed and bank material sizes in each quadrant and sample.

12.5 Results

Examples of results are presented for the discharge measurements, cross-sections, habitat maps, sediment size analysis, macro invertebrate sampling and vegetation sampling. Cross-sectional profile change is addressed in Section 12.5.1, habitat and hydraulic biotope change in Section 12.5.2, river-bed particle size distribution in Section 12.5.3, macro invertebrates in Section 12.5.4 and vegetation pattern change and hydrogeomorphology in Section 12.5.5. The discharge results provide input to Sections 12.5.1, 12.5.2 and 12.5.5.

12.5.1 Cross-sectional profile change

The Western Cape had exceptionally high rainfall from May to September 2001. This is indicated in Figure 12.2. When examining the hydraulics (Table 12.8) and cross-sectional data (Figure 12.5a-j), it is clear that the particularly large flooding events of 2001 created high velocities that were capable of moving coarse bed material. Discharge measurements were very variable with the different hydraulic biotopes having different readings although the same amount of water was flowing through the channel. Different readings were also obtained between the same hydraulic biotopes, for example readings taken on 13 and 28 July 2001 (Table 12.8). It is clear that biotopes differ in their substratum and flow composition at any one time and that this had an affect on the depth and flow velocities measured and thus calculated discharges. At Site 1 the largest flood observed was at the beginning of July 2001, the 13th, which produced an averaged velocity of 0.961 m s^{-1}.

Table 12.8 Channel width, depth, velocity and discharge for Site 1 on the Lourens River between May and September 2001.

Date	Cross-Section (Dominant Hydraulic Biotope)	Channel Width (m)	Average depth (Range) (m)	Average Velocity (Range) (m s^{-1})	Discharge (m^3 s^{-1})	Average Discharge (m^3 s^{-1}) (Standard deviation)
10 May 01	1.2 (Run)	10.4	0.21 (0.09-0.32)	0.353 (0.001-0538)	0.676	0.76 (0.12)
	1.8 (Run)	9.1	0.24 (0.15-0.29)	0.392 (0.015-0.571)	0.851	
12 May 01	1.2 (Run)	8.9	0.40 (0.11-0.50)	0.144 (0.001-0.353)	0.408	0.36 (0.07)
	1.8 (Run)	5.6	0.25 (0.10-0.38)	0.216 (0.001-0.406)	0.309	
29 Jun 01	1.2 (Riffle)	9.6	0.37 (0.10-0.54)	0.214 (0.015-0.511)	0.818	0.53 (0.27)
	1.5 (Riffle)	6.3	0.16 (0.14-0.26)	0.490 (0.035-0.849)	0.473	
	1.8 (Riffle)	5.8	0.35 (0.20-0.45)	0.153 (0.001-0.306)	0.297	
13 July 01	1.2 (Riffle)	10.6	0.49 (0.26-0.76)	0.725 (0.124-1.773)	4.042	4.28 (0.21)
	1.5 (Riffle)	8.6	0.50 (0.13-0.54)	1.043 (0.832-1.405)	4.377	
	1.8 (Riffle)	8.0	0.50 (0.18-0.68)	1.114 (0.788-1.480)	4.423	
28 July 01	1.2 (Riffle)	6.7	0.64 (0.30-0.88)	0.470 (0.114-1.134)	2.311	1.99 (0.36)
	1.5 (Riffle)	9.6	0.38 (0.25-0.53)	0.532 (0.001-0.072)	2.064	
	1.8 (Riffle)	8.0	0.35 (0.13-0.43)	0.578 (0.022-1.001)	1.608	
18 Sept 01	1.2 (Run)	8.8	0.62 (0.15-1.00)	0.298 (0.101-0.703)	2.150	0.00 (0.00)

Marked changes in channel geometry (width, and depth) were observed between June 2001 and March 2002 at Site 1. A distinction can be made however, between the upper and lower section of Site 1 with respect to changes in channel morphology. It was within the upper section (Figure 12.5; cross-profiles 1.1-1.5) that changes were most visible whereas the lower section (cross-profiles 1.6-1.9) showed very little significant change in bed and bank profiles with increased discharge. However, it is evident from the combined cross-sectional profiles in the upper section and variable discharge results, that the changes observed were not simply the result of a single high flow event, but represented the cumulative effect of the wide range of discharges experienced in 2001.

Combined cross-sections 1.3, 1.4 and 1.5 (Figure 12.5c, d and e) show channel widening due to severe erosion and retreat of the left bank. The latter receded by 5-7m. Repeat surveys in June 2001, August 2001 and March 2002, revealed channel deepening at cross-sections 1.2, 1.4 and 1.5 and deposition of sediments in the channel at cross-section 1.1 and to the right of the thalweg at cross-sections 1.1, 1.3, 1.4 and 1.5. An average of 0.7 m of bed fill was deposited during a moderate flood in August 2001 at cross section 1.1, with sediment supplied during the flood by observed upstream

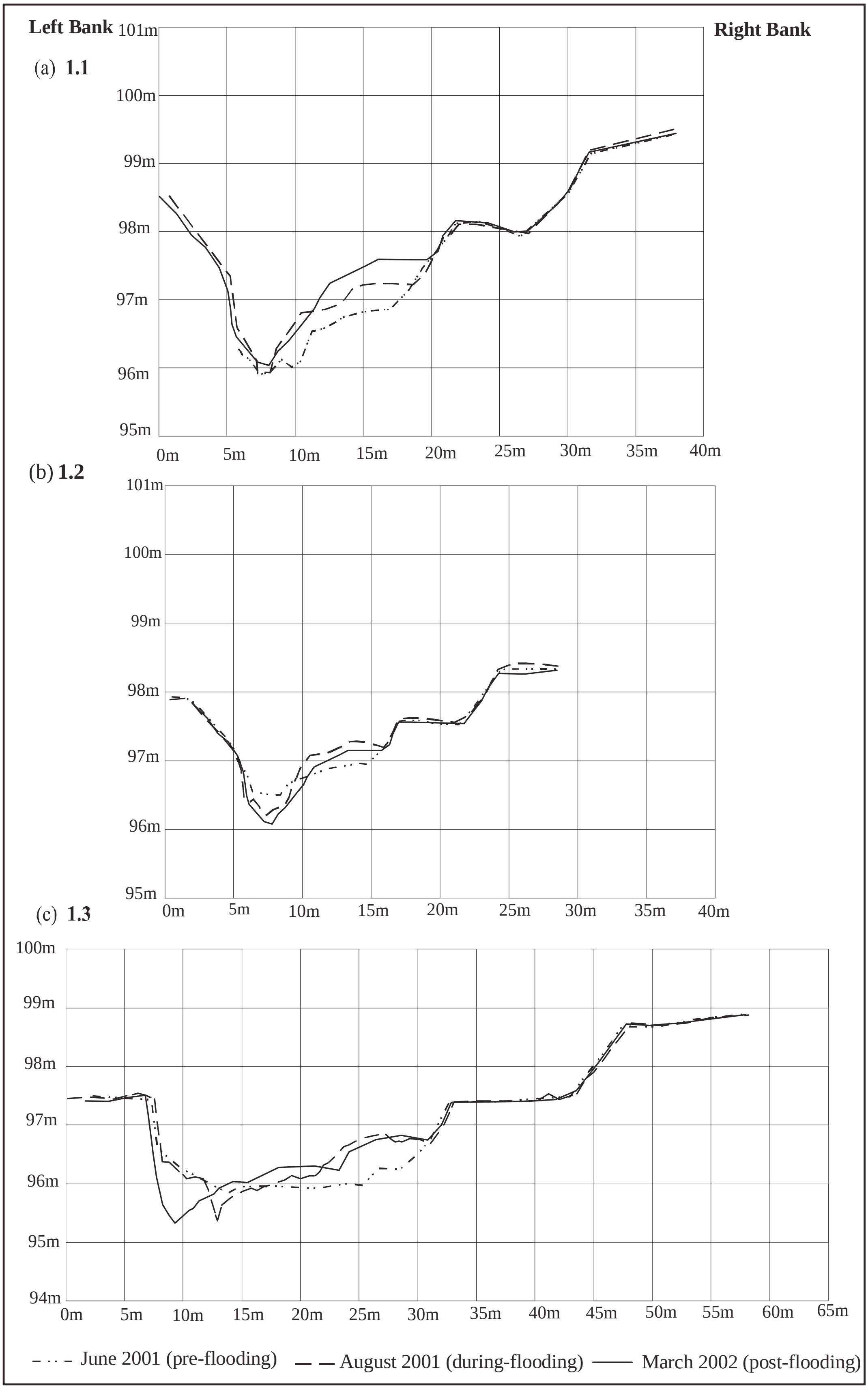


Figure 12.5 a-c Cross-section profiles 1.1, 1.2 and 1.3.

Figure 12.5 d-f Cross-section profiles 1.4, 1.5 and 1.6.

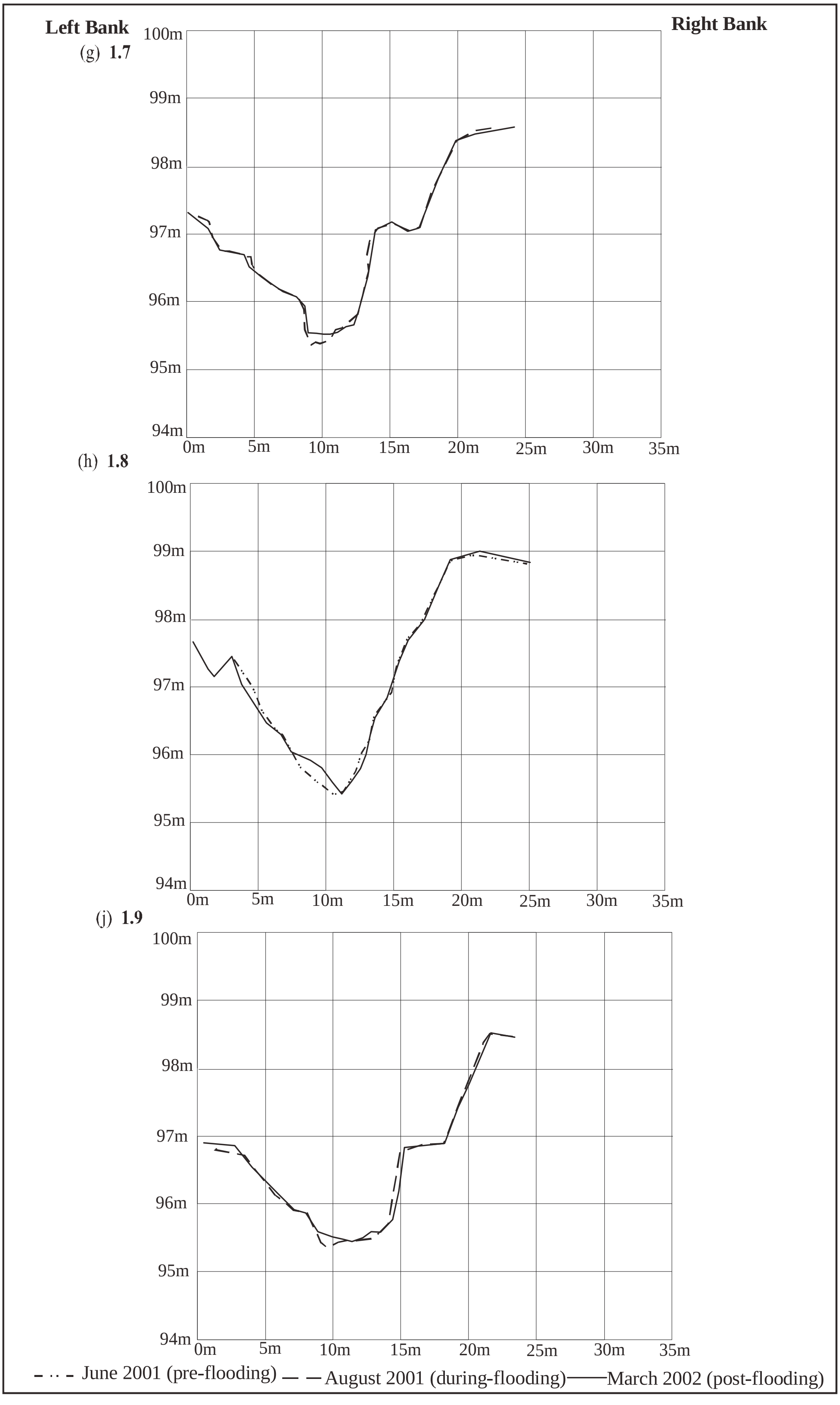


Figure 12.5 g-j Cross-section profiles 1.7, 1.8 and 1.9.

bank erosion. On the upstream right bank undercutting took place behind the gabion mattresses while deposition of 0.5 m of sediment occurred at the adjacent right hand lateral bar (active channel bank). It is evident from the profiles surveyed in August 2001 and March 2002 at cross-section 1.2 (Figure 12.5b), that scouring of about 0.3 m resulted in a lowering of the channel bed and adjacent lateral bar compared to its position in August 2001. From the above it is evident that the upper section of Site 1 underwent both deepening of the channel as well as a build-up of areas adjacent to the main channel.

In contrast, the down-stream sections (cross-sections 1.6-1.9), were more stable with little change in channel morphology. The only small but perceptible change was a 0.15m to 0.25m build-up of the channel at all the cross-sections as well as the development of a small lateral bar at channel cross-profile 1.6 (Figure 12.5f). It is assumed that since the profile did not indicate any erosion on the bed or banks, the materials were supplied from the upstream cross-sections 1.2 to 1.5.

12.5.2 Habitat and hydraulic biotope change

Substratum maps

Significant changes in the substratum-flow habitat maps were observed as a response to the particularly high rainfall in 2001 (Figure 12.2). The habitat maps of this section provided information on the distributions and proportions of the different substrata (Figure 12.6a-c and Table 12.9) and flow types with a change in discharge from the beginning to the end of the study period. A comparison of the substratum and flow type habitat maps indicated the observed changes during the autumn and winter of 2001, and the summer of 2002. Categories of classes used in substratum mapping are described in Tables 12.3-12.5. The distribution pattern of the three habitat maps varied considerably over the study period, indicating that the heavy rainfall in 2001 clearly played a role in the results.

At the beginning of the study period, the habitat map of April 2001 revealed that the site was predominately characterized by a mixed cobble, gravel, fine and algae (C+G+F+A) substratum and a mixed cobble, gravel, fine, boulder and algae (C+G+F+B+A)substratum. There were areas of marginal vegetation with or without sand and gravel, particularly in the upper and lower sections of the site. Small areas of mixed categories, such as an island located immediately downstream of cross-section 1.3, also occurred at the site. Overall the mixed cobble-gravel accounted for 61% of the wetted area, marginal vegetation with or without sand and gravel 22% and the small areas of mixed categories 17%.

Figure 12.6b showed the substrata distribution pattern of the same site during the floods in August 2001. The results displayed that the site was mainly characterized by a mixed substratum of gravel, cobble, boulder, fines and algae (G+C+B+F+A), a mixed cobble, gravel, fines, boulder and algae (C+G+F+B+A) substratum and fines with gravel substratum (F+G). The fines with gravel and the mixed gravel-cobble substratum were particularly apparent in the upstream end of the site while the

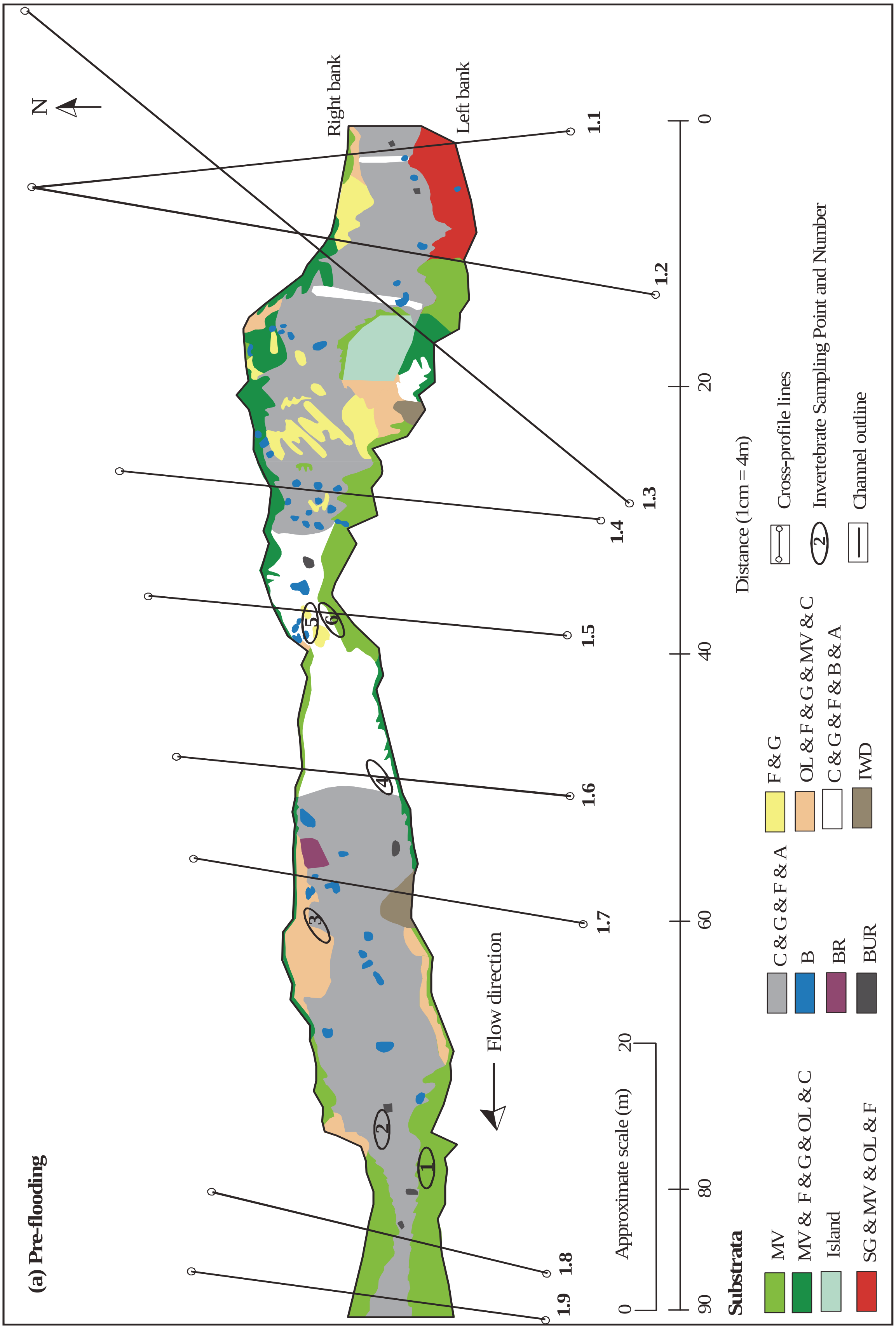


Figure 12.6a **Substrata of Site 1, Lourens River (April 2001).**

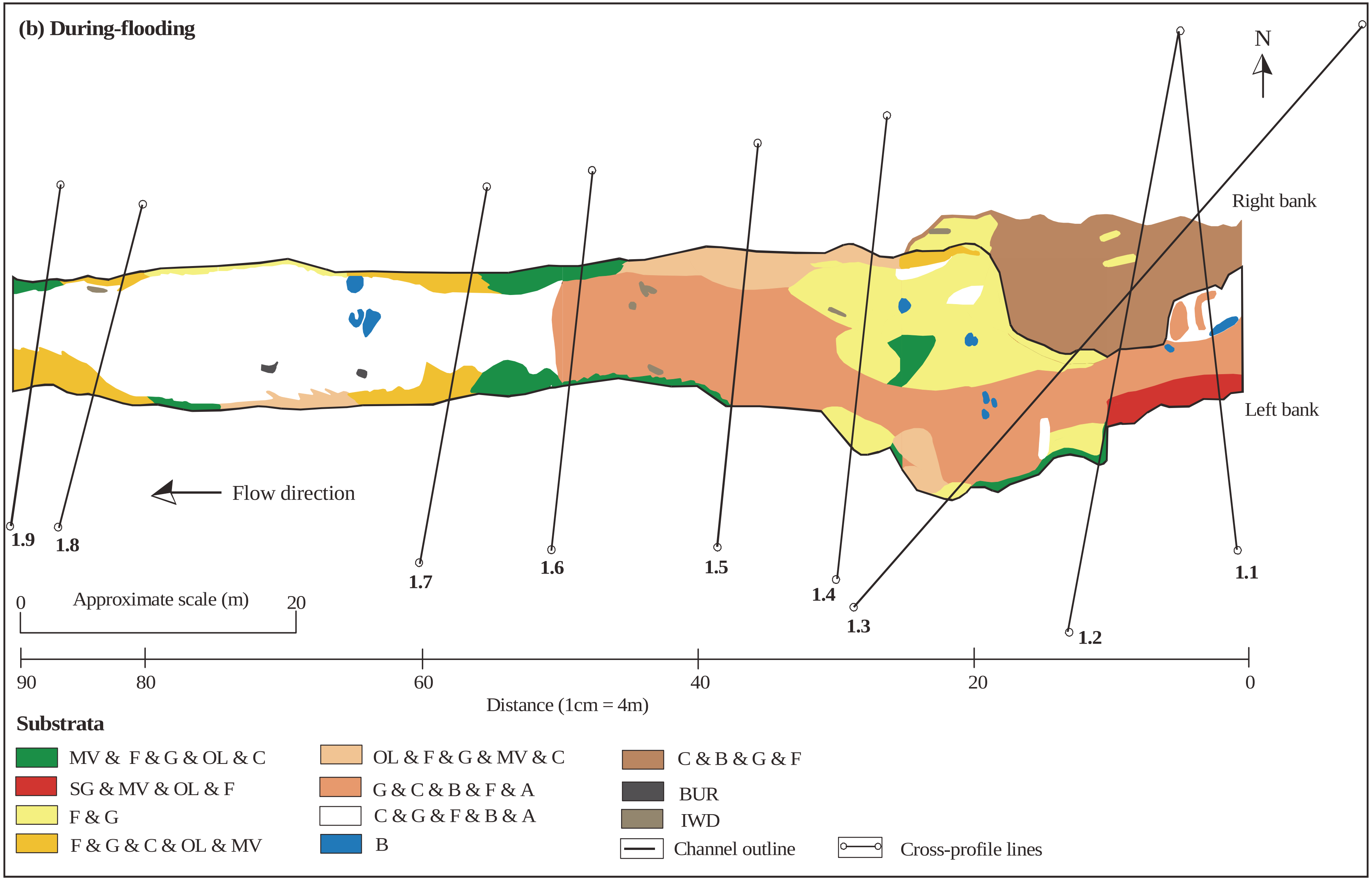


Figure 12.6b **Substrata of Site 1, Lourens River (August 2001).**

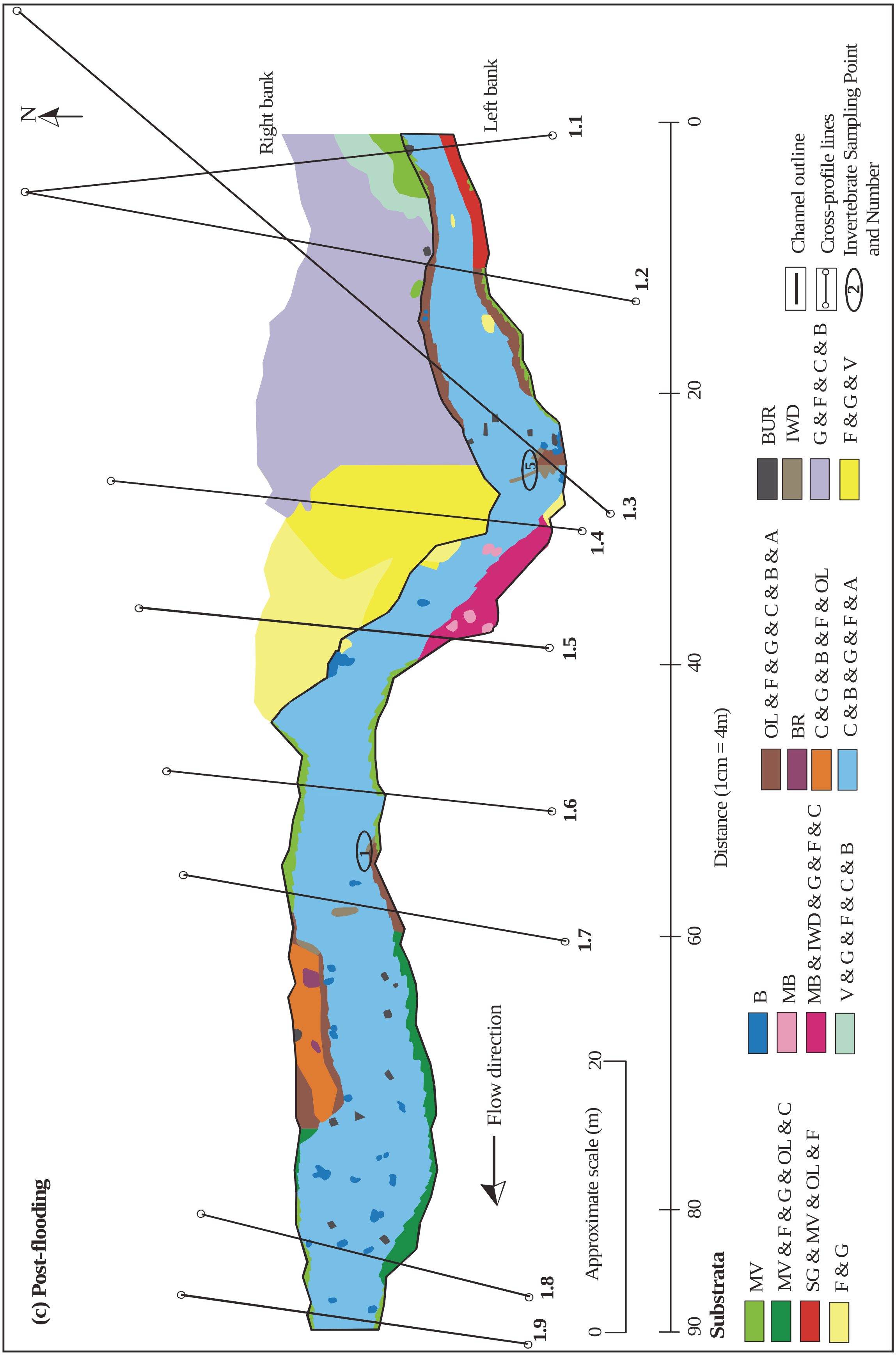


Figure 12.6c Substrata of Site 1, Lourens River (February 2002).

mixed cobble-gravel substratum was located in the downstream end of the site. Small areas of marginal vegetation mixed with different substrata and organic litter, also occurred at the site. A lateral bar characterized by a mixed cobble-gravel substratum developed on the adjacent right bank (discussed in Section 12.5.1). The bar, which was vegetated, occupied an area of 161 m^2. Overall the mixed gravel-cobble substrata accounted for 31% of the wetted area, the mixed cobble-gravel 36%, while the fines with gravel accounted for 14% and the small area categories 19%.

The substratum habitat map of February 2002, after the 2001 flooding events (Figure 12.6c) showed that the site was predominately characterized by a mixed cobble, boulder, gravel, fines and algae (C+B+G+F+A) substratum. There were also small areas of marginal vegetation with or without sand and gravel, organic litter with mixed substrata as well as a new category referred to as mud blocks with or without debris and mixed substrata. Bank erosion at cross sections 1.3 and 1.4 on the left bank exposed the mud blocks, a more resistant type of slope material at the bottom of the bank. The exposed mud blocks were located between the two cross-sections referred to. Patches of building rubble and boulders were found throughout the whole active channel.

Overall, the mixed cobble, boulder, gravel, fines and algae (C+B+G+F+A) substratum category accounted for 60% of the wetted area, organic litter with mixed substrata 11%, marginal vegetation 5%, mud block with in-channel woody debris and mixed substrata 7% and the small areas category 17%. Continued sedimentation resulted in enlargement of the lateral bar, occupying a 459.8 m^2 area.
A comparison of the three habitat maps for Site 1 clearly displayed the fact that the substrata of the wetted channel were reworked and redistributed. It also communicated the change in the adjacent lateral bar as well as the observed channel migration reported in section 12.5.1. The results for the period April 2001 to March 2002 presented above have suggested the following:

- transition from a mixed cobble, gravel, fine and algae substratum (Figure 12.6a) to a mixed cobble, boulder, gravel, fines and algae substratum (Figure 12.6c);
- fining of the substratum of the lateral bar;
- gradual migration of the wetted channel towards the left bank floodplain, with a sinuous channel pattern emerging.

Flow maps

The results of the flow type maps are presented in Table 12.10 and Figure 12.7a-c. Flow type categories are summarized in Table 12.6. It is evident from the maps and data in Table 12.8 that most changes in flow type took place in the winter of 2001 when channel discharge was high.

Table 12.9 Substratum cover percentage and area at Site 1, Lourens River, in April 2001, August 2001 and February 2002.

Period	**Pre-flooding**		**During-flooding**		**Post-flooding**	
Date	**10 Apr 2001**		**1 Aug 2001**		**22 Feb 2002**	
Substrata	**Area (m^2)**	**%**	**Area (m^2)**	**%**	**Area (m^2)**	**%**
MV	112.3	17.0	0.0	0.0	26.0	5.0
MV+F+G+OL+C	33.9	5.0	37.5	4.0	18.4	4.0
V+G+F+C+B--	17.1	3.0	0.0	0.0	0.0	0.0
SG+MV+OL+F	22.9	4.0	16.9	2.0	9.1	2.0
F+G	27.0	4.0	123.1	14.0	5.4	1.0
F+G+C+OL+MV	0.0	0.0	41.8	5.0	0.0	0.0
OL+F+G+C+B+A	0.0	0.0	0.0	0.0	54.8	11.0
OL+F+G+MV+C	18.1	3.0	64.9	7.0	0.0	0.0
G+C+B+F+A	0.0	0.0	263.8	31.0	0.0	0.0
C+B+G+F+A	0.0	0.0	0.0	0.0	307.5	60.0
C+G+F+B+A	91.3	14.0	306.7	36.0	0.0	0.0
C+G+F+A	309.4	47.0	0.0	0.0	0.0	0.0
B	14.0	2.0	6.5	1.0	10.95	2.0
MB	0.0	0.0	0.0	0.0	3.4	1.0
MB+IWD+G+F+C	0.0	0.0	0.0	0.0	33.8	7.0
BR	0.8	0.0	0.0	0.0	18.0	4.0
BUR	1.4	0.0	0.6	0.1	3.75	1.0
IWD	7.8	1.0	0.8	0.1	4.0	1.0
C+B+G+F**	0.0	0.0	147.0	91.0	0.0	0.0
F+G**	0.0	0.0	14.0	9.0	0.0	0.0
C+G+B+F+OL**	0.0	0.0	0.0	0.0	21.1	5.0
G+F+C+B**	0.0	0.0	0.0	0.0	254.4	55.0
F+G+V**	0.0	0.0	0.0	0.0	91.3	20.0
V+G+F+C+B**	0.0	0.0	0.0	0.0	15.9	3.0
MV**	0.0	0.0	0.0	0.0	8.1	2.0
F+G**	0.0	0.0	0.0	0.0	69.0	15.0
Total mapped area (m^2)	656.0	100.0	862.6	100.0	516.2	100.0

Note: ** = Lateral bar area excluded from total mapped area for August 2001 and February 2002
-- = Island

Table 12.10 Flow type area and percentage at Site 1, Lourens River on 9 April 2001, 27 July 2001 and 1 February 2002.

Flow type	Area (m^2) of flow types			Flow types cover percentages (%)		
	Pre-flooding	During-flooding	Post-flooding	Pre-flooding	During-flooding	Post-flooding
Date	**Apr 2001**	**Jul 2001**	**Feb 2002**	**Apr 2001**	**Jul 2001**	**Feb 2002**
NF (No flow)	36.9	10.3	40.0	5.0	2.0	7.0
BPF (Barely percetible flow)	98.5	13.3	11.5	13.0	2.0	2.0
SMF (Smooth Flow)	279.1	115.1	372.3	39.0	13.0	65.0
MRF (Medium rippled flow)	151.4	333.9	100.8	21.0	38.0	17.0
FTF (Fast turbulent flow)	157.7	418.3	49.7	22.0	47.0	9.0
Total wetted area (m^2)	686.7	880.6	534.3	100.0	100.0	100.0

The flow type maps displayed a typical summer and winter situation with reference to the distribution of flow patterns. Summer seasons (Figure 12.7a and c) were characterized by a dominance of the NF (no flow), BPF (barely perceptible flow) and SMF (smooth flow) categories, while MRF (medium rippled flow) and FTF (fast turbulent flow) dominated the winter pattern (Figure 12.7b). This is also well illustrated in Table 12.10 in the percentage area covered by the different flow types. The smooth flow category (SMF), for example, covered an area of approximately 39%-65% during the summer months at low baseflow conditions and only about 13% during the winter. In a similar way the fast turbulent flow category (FTF) covered an area of between 9% and 22 % in summer and approximately 47% in the winter, during high flow conditions. The high energy of the flow conditions was further illustrated by the total disappearance of the island on the last flow map. The island was totally exposed in April 2001 (Figure 12.7a), completely inundated during the winter in July 2001 (Figure 12.7b) and absent in February 2002 (Figure 12.7c).

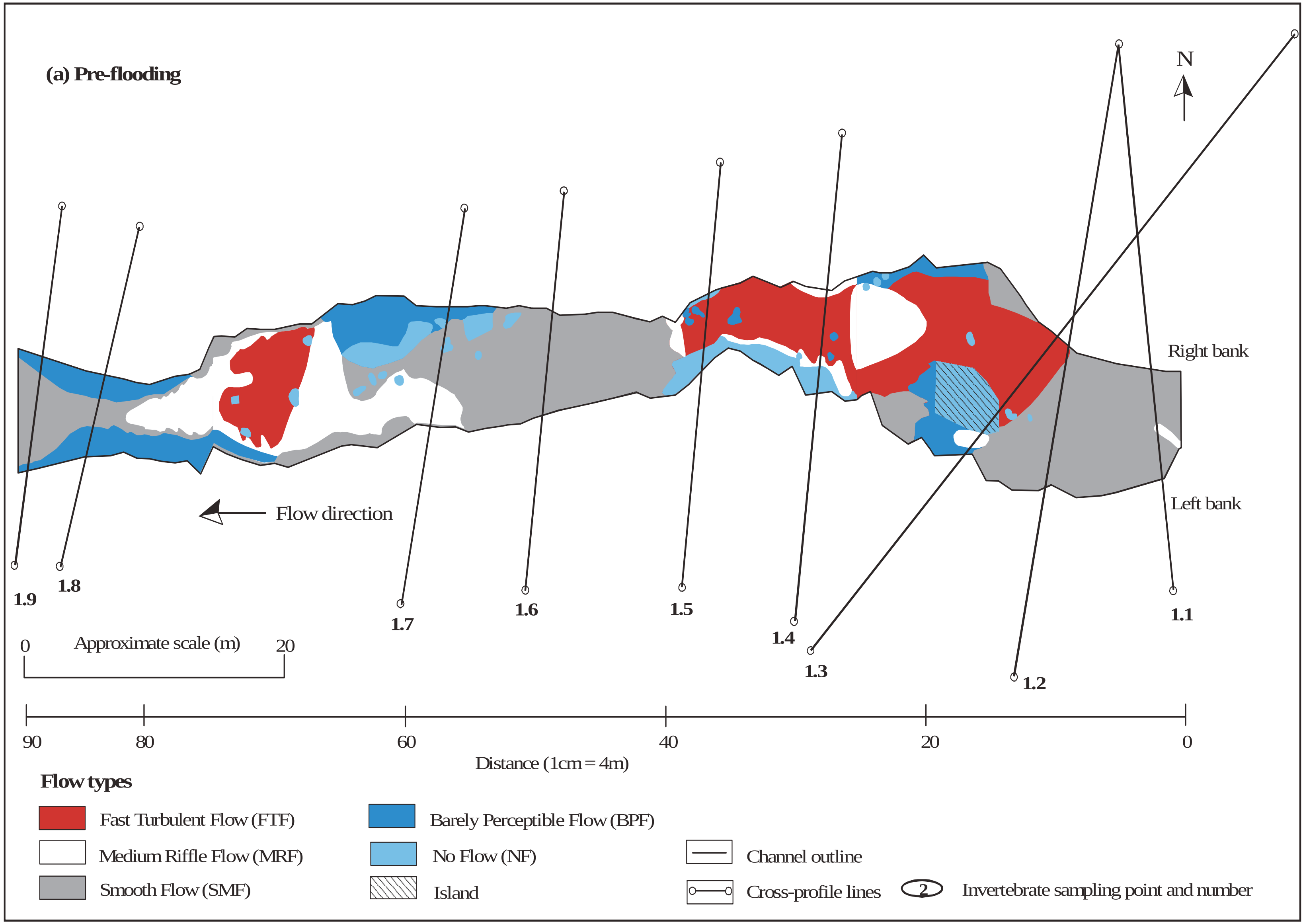


Figure 12.7a Flow types of Site 1, Lourens River (April 2001).

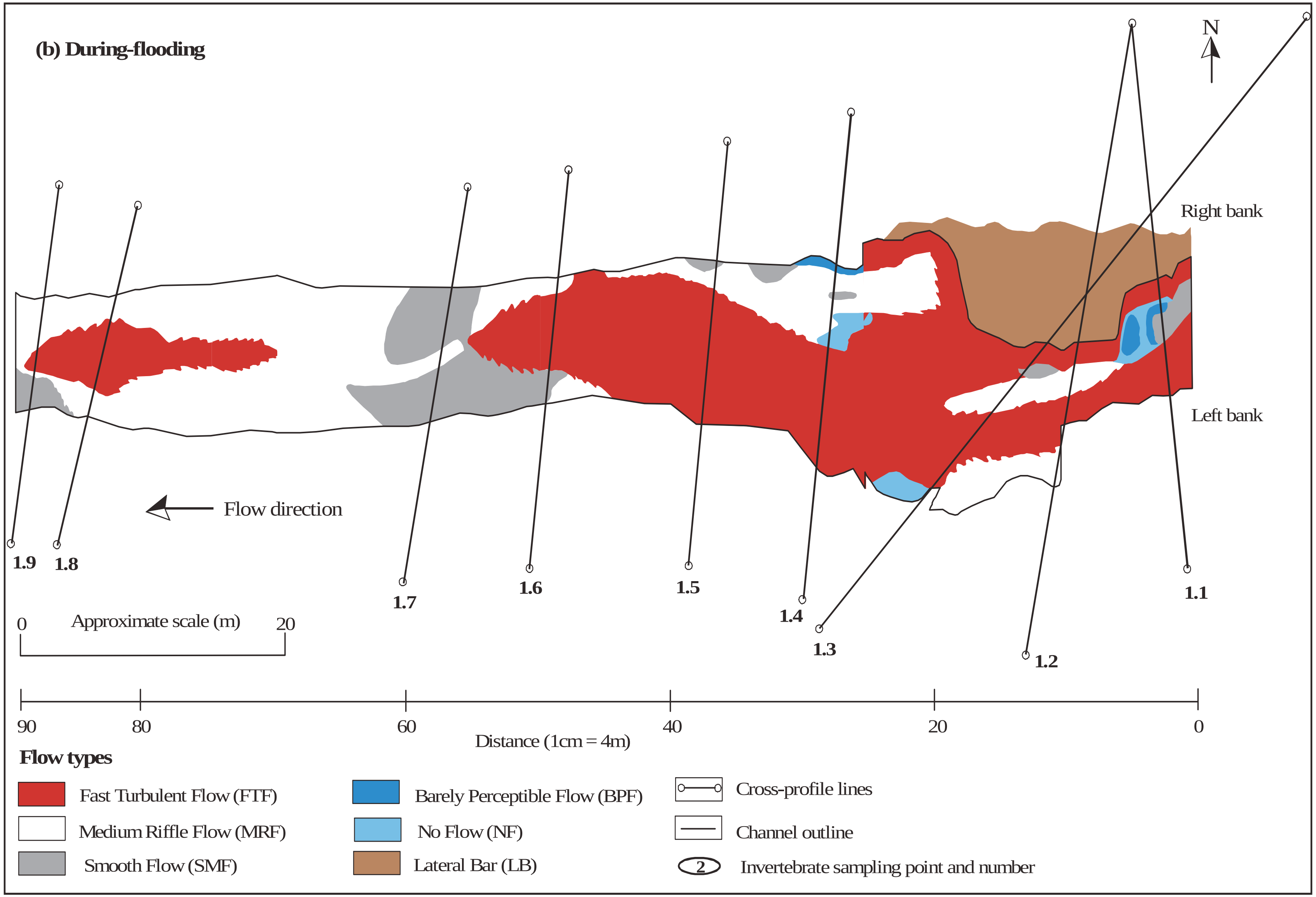


Figure 12.7b Flow types of Site 1, Lourens River (July 2001).

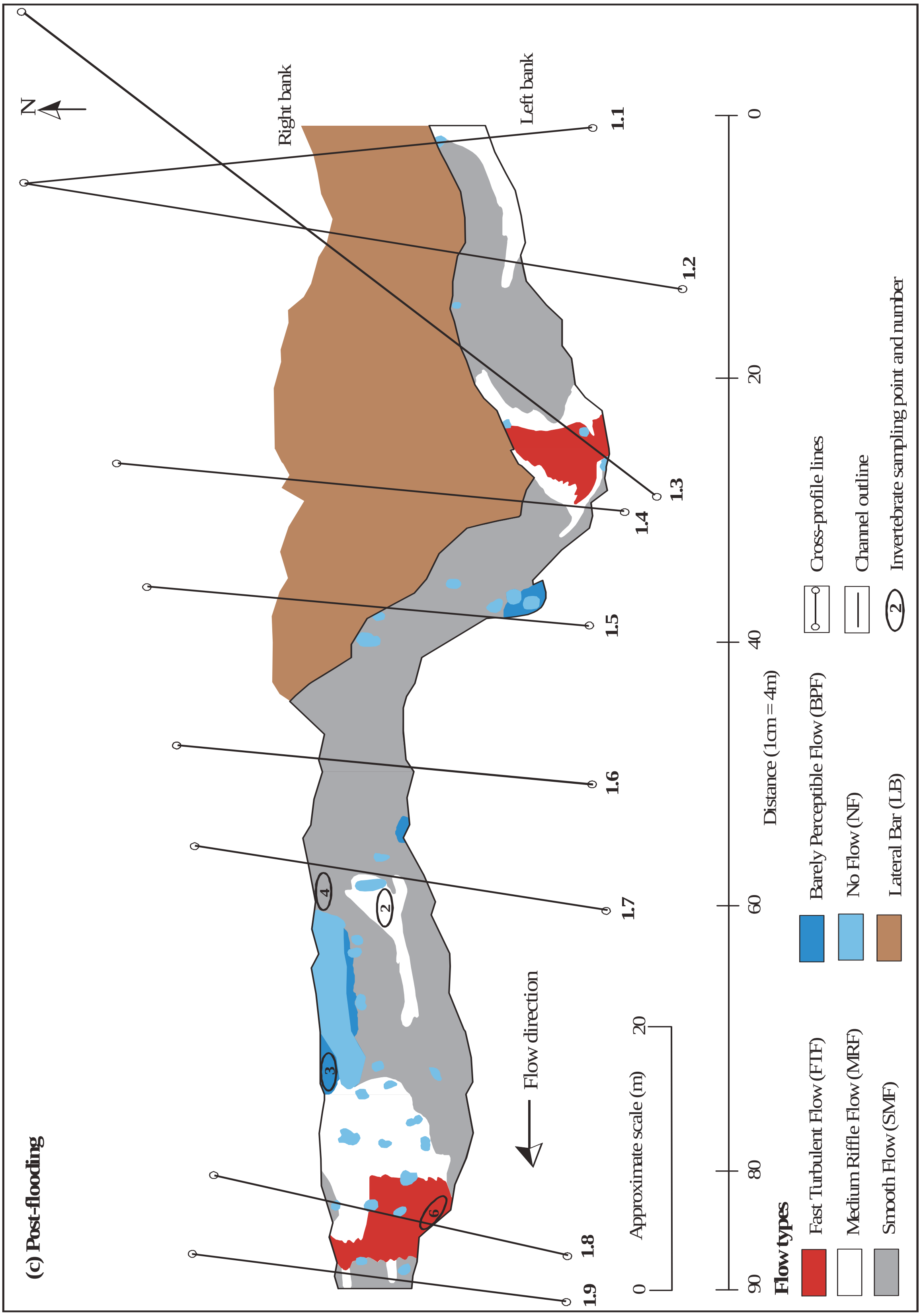


Figure 12.7c Flow types of Site 1, Lourens River (February 2002).

Hydraulic biotope change

Flow types used to define hydraulic biotopes are shown in Table 12.6. Biotopes (Table 12.1) can be conceptualized as vertical cells within the flow. The individual sediments underlying the biotopes can also form local hydraulic controls resulting in a mosaic of hydraulic biotopes. Hydraulic biotopes were observed and classified in the field using the concepts and ideas formalized by Rowntree & Wadeson (1999). The hydraulic biotopes for Site 1 were reclassified along each cross-section mainly with the mapping of the flow type habitat maps and their associated hydraulic conditions in April 2001, July 2001 and February 2002. Photographic evidence aided the classification / re-classification of hydraulic biotopes. The transformation of dominant hydraulic biotopes from one class to another in response to changing discharge for Site 1 can be seen in Table 12.8 and Table 12.11.

Discharge readings for July 2001 reached values of up to 4.4 m^3s^{-1} and velocities of up to 1.77 m s^{-1} (Table 12.8). Flow types during this period were mainly MRF (medium rippled flow) and FTF (fast turbulent flow) (Table 12.11). During the period April 2001 to July 2001 hydraulic biotopes at cross-sections 1.1, 1.2, 1.4 and 1.8 changed from runs to riffles (Table 12.11) in response to higher discharge (Table 12.8). Field classification of hydraulic biotopes placed cross-section 1.7 as a run at all discharges while cross-sections 1.3 and 1.5 remained riffles. From July 2001 to February 2002 hydraulic biotopes at cross-sections 1.2 and 1.4 changed back from riffles to runs while those at cross-sections 1.1 and 1.8 remained riffles. Although no discharge readings for February 2002 were available, it can be assumed that low baseflow conditions typical of the summer prevailed. When the substrata along these cross-sections are compared (Figure 12.6b and c), it is clear that where hydraulic biotopes along cross-sections 1.2 and 1.4 changed from riffles back to runs, a substratum change from fine to coarse material occurred. In the case of the hydraulic biotope at cross-sections 1.1 and 1.8 that remained the same, no such change in substratum occurred. From the results outlined above, it is probably true to say that the higher winter discharge lead to a change in the distribution of substrata and this had an influence on the observed biotope change.

12.5.3 Gravel-bed river particle size distribution

In this study the particle size distribution of bed and bank material for Site 1 was done at cross-section 1.3 (Figure 12.8) to verify that the sizes visually estimated and used in the substratum habitat mapping exercise, Section 12.5.2, were accurate. Particle size analysis was furthermore undertaken mainly to use in a descriptive capacity. The results were intended to indicate possible links between substratum distribution and its connection with changes in channel morphology and habitats. Categories of substratum are given in Table 12.3. Measurements and samples of particle sizes were taken in August of 2001. Cross-section 1.3 represented a riffle hydraulic biotope class in 2001 and was chosen because it represented most of the available particle size classes that were observed at Site 1.

Table 12.11 Short-term hydraulic biotope change in Site 1 along cross-sections 1.1 to 1.5, 1.7 and 1.8 during the study period. Flow types as in Table 12.6.

Date	**Cross-section number**	**Flow type along cross-section**	**Dominant Hydraulic Biotope**
April 2001	1.1	SMF	run
April 2001	1.2	SMF	run
April 2001	1.3	FTF and SMF	riffle
April 2001	1.4	FTF, MRF and NF	run
April 2001	1.5	FTF, BPF and NF	riffle
April 2001	1.7	BPF, SMF and MRF	run
April 2001	1.8	BPF and SMF	run
June 2001	1.1	FTF	riffle
June 2001	1.2	MRF	run
June 2001	1.3	FTF, SMF and MRF	riffle
June 2001	1.4	MRF	run
June 2001	1.5	FTF and MRF	riffle
June 2001	1.7	SMF and MRF	run
June 2001	1.8	MRF	run
July 2001	1.1	FTF	riffle
July 2001	1.2	FTF and MRF	riffle
July 2001	1.3	FTF and MRF	riffle
July 2001	1.4	FTF and MRF	riffle
July 2001	1.5	FTF and MRF	riffle
July 2001	1.7	SMF and MRF	run
July 2001	1.8	FTF and MRF	riffle
February 2002	1.1	SMF and MRF	riffle
February 2002	1.2	SMF and MRF	run
February 2002	1.3	SMF, MRF and FTF	riffle
February 2002	1.4	SMF	run
February 2002	1.5	SMF	riffle
February 2002	1.7	SMF	run
February 2002	1.8	FTF and MRF	riffle

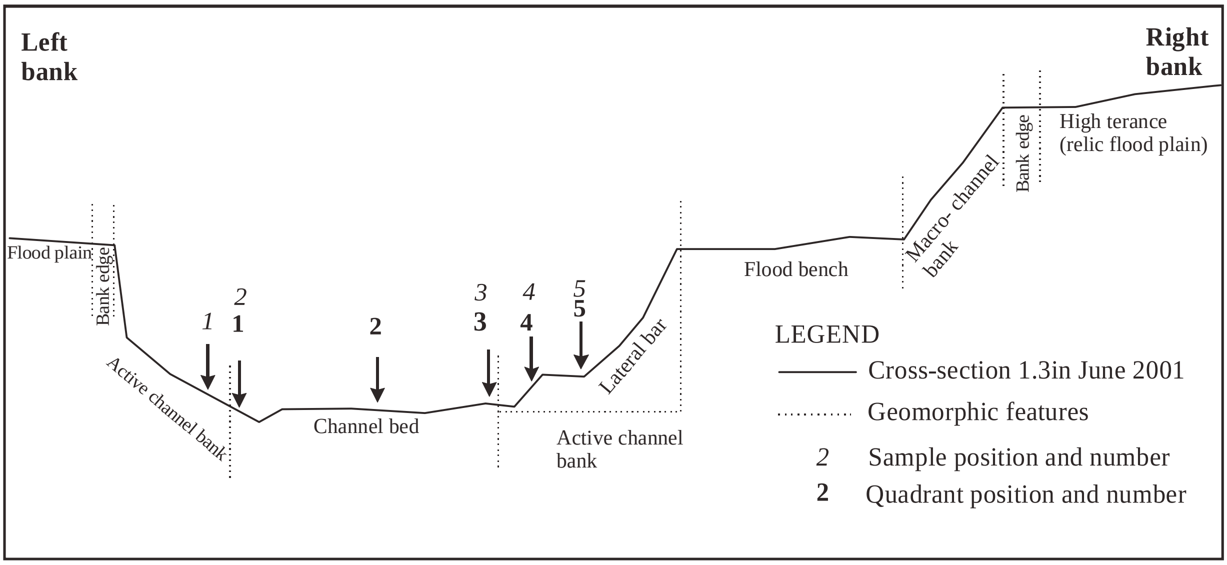


Figure 12.8 Substratum quadrant and sample positions at Site 1, cross-section 1.3, in 2001.

An overall summary of the substrata along cross-section 1.3 for 2001 is provided in Tables 12.12 and 12.13. Figures 12.9a showed the cumulative frequency distribution of coarse fractions (cobbles and gravel) from the quadrants along profile 1.3 while Figure 12.9b showed the cumulative curves for the sand, silt and clay fractions from the samples taken in each quadrant. According to Boggs (1995) a measure of the range of grain sizes present and the magnitude of the spread of the sizes around the mean size in a sediment population is defined as sorting. The slopes of the cumulative curves reflect the sorting of the sample. Gentle slopes indicate poor sorting whereas steep slopes indicate good sorting. Mean grain size and sorting values were also calculated from the cumulative curves (Figure 12.9a and 12.9b) using the Folk & Ward (1957) formulas displayed in Table 12.14.

According to Table 12.15 all five quadrants were dominated by very coarse material. Mean values varied from -4.5 ø to -5.2 ø (ø = $-\log_2 d_i$ where d_i is the diameter of the particle in millimetre). High negative values indicate coarse material in the gravel to large cobble category. Sorting values for the quadrants varied between 1.09 and 1.47 indicating poor sorting (Folk & Ward 1957). Mean values for samples one to five all fall within the fine to coarse sand fraction with sorting values ranging from 0.78 to 1.1, moderately to poorly sorted. From the above analysis it can be seen that the left bank sample (sample one) had a fine sand composition, while those from the right hand active channel bank were coarse. The size distribution in sample one (Table 12.13) indicated that the left bank had a high percentage of medium to fine sand (91%), low silt (8%) and almost no clay (< 1%). The left bank therefore consisted of material that could lead to bank instability especially in the absence of stabilising vegetation.

Table 12.12 **Frequency and cumulative frequency of stones along cross-section 1.3 for Site 1, in quadrants 1 to 5.** Quadrant codes: 1.3-1-01, represents the following: 1 = site #, 3 = cross-section #, 1 = quadrant # and 01 = year.

Quadrant code		1.3-1-01		1.3-2-01		1.3-3-01		1.3-4-01		1.3-5-01	
Grain size class	**Size class (mm)**	**Frequency**	**Cumulative frequency**	**Frequency**	**Cumulative frequency**	**Frequency**	**Cumulative frequency**	**Frequency**	**Cumulative frequency**	**Frequency**	**Cumulative frequency**
Small gravel	8-16	39	39	25	25	19	19	26	26	24	24
Medium gravel	16-32	27	66	13	38	16	35	35	61	28	52
Large gravel	32-64	22	88	26	64	40	75	23	84	25	77
Small cobble	64-128	10	98	24	88	21	96	16	100	21	98
Large cobble	128-256	2	100	12	100	4	100	0	0	2	100
Boulder	>256	0	0	0	0	0	0	0	0	0	0
Building rubble	200-300	0	0	0	0	0	0	0	0	0	0

Table 12.13 **Sieve analysis for samples 1-5, Site 1, Lourens River August 2001.** Sample codes: 1.3-*1*-01, represents the following: 1 = site #, 3 = cross-section #, *1* = sample # and 01 = year.

Sample code		1.3-*1*-01		1.3-*2*-01		1.3-*3*-01		1.3-*4*-01		1.3-*5*-01	
Size class name	**Grain size (ø)**	**Mass %**	**Cumulative mass %**	**Mass %**	**Cumulative mass %**	**Mass %**	**Cumulative mass %**	**Mass %**	**Cumulative mass %**	**Mass %**	**Cumulative mass %**
Coarse	-0.50	0.98	0.98	68.47	68.47	18.80	18.80	21.74	21.74	45.59	45.59
Sand	0.00	0.79	1.77	7.51	75.98	3.42	22.22	13.69	35.43	10.18	55.77
	0.25	0.69	2.46	3.32	79.30	3.31	25.53	8.45	43.88	5.63	61.40
	0.75	2.60	5.06	5.90	85.02	11.30	36.83	18.03	61.91	11.48	72.88
Medium	1.25	7.44	12.50	6.05	91.25	19.96	56.79	17.79	79.70	11.83	84.71
Sand	1.75	17.04	29.54	4.52	95.77	23.09	79.88	12.01	91.71	8.31	93.02
	2.25	25.71	55.25	2.32	98.09	13.67	93.55	5.54	97.25	3.96	96.98
	2.75	23.36	78.61	0.93	99.02	4.78	98.33	1.53	98.78	2.12	99.10
Fine	3.25	7.70	86.31	0.68	99.70	0.99	99.32	0.33	99.11	0.44	99.54
Sand	3.75	4.50	90.81	0.15	99.85	0.37	99.69	0.77	99.88	0.20	99.74
Silt	4.25-8	8.37	99.18	0.03	99.88	0.05	99.74	0.02	99.90	0.01	99.75
Pan amount	0.00	0.00	100.00	0.12	100.00	0.26	100.00	0.11	100.00	0.23	100.00
Clay	9.00	0.82	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

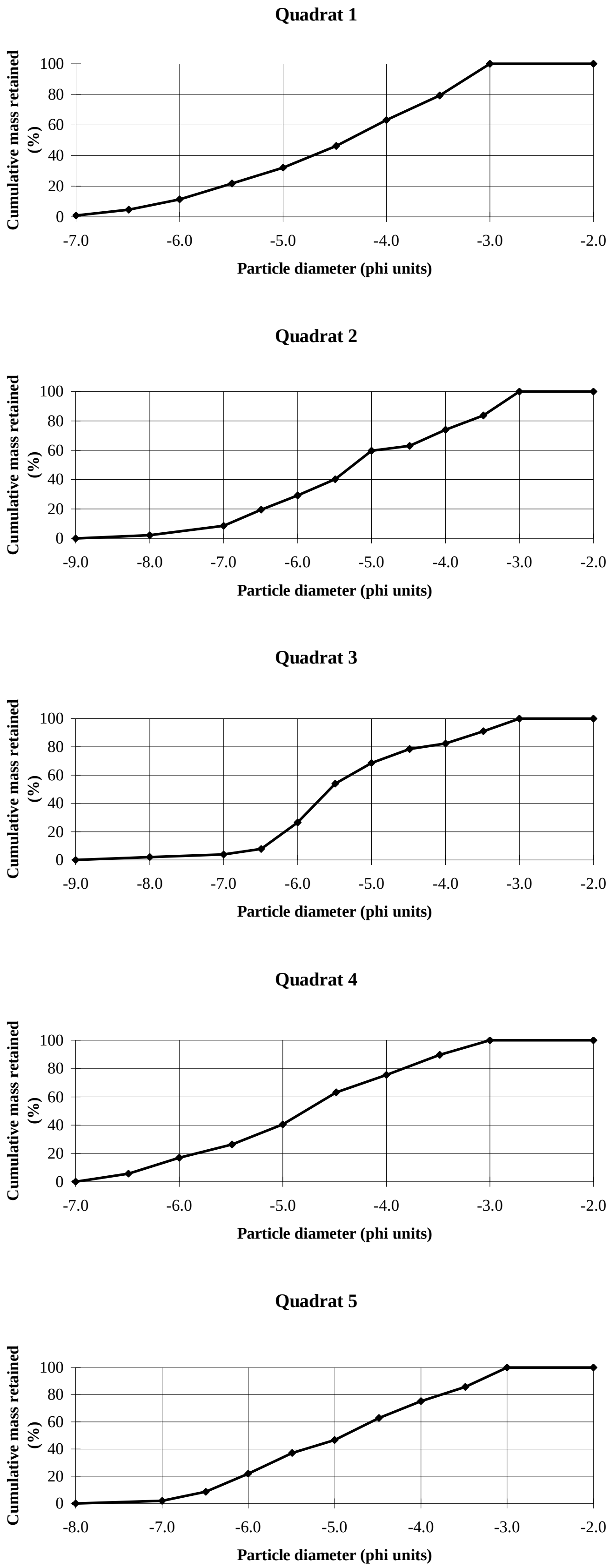


Figure 12.9a Grain size cumulative curves for quadrats 1-5 at cross-section 1.3, Site 1.

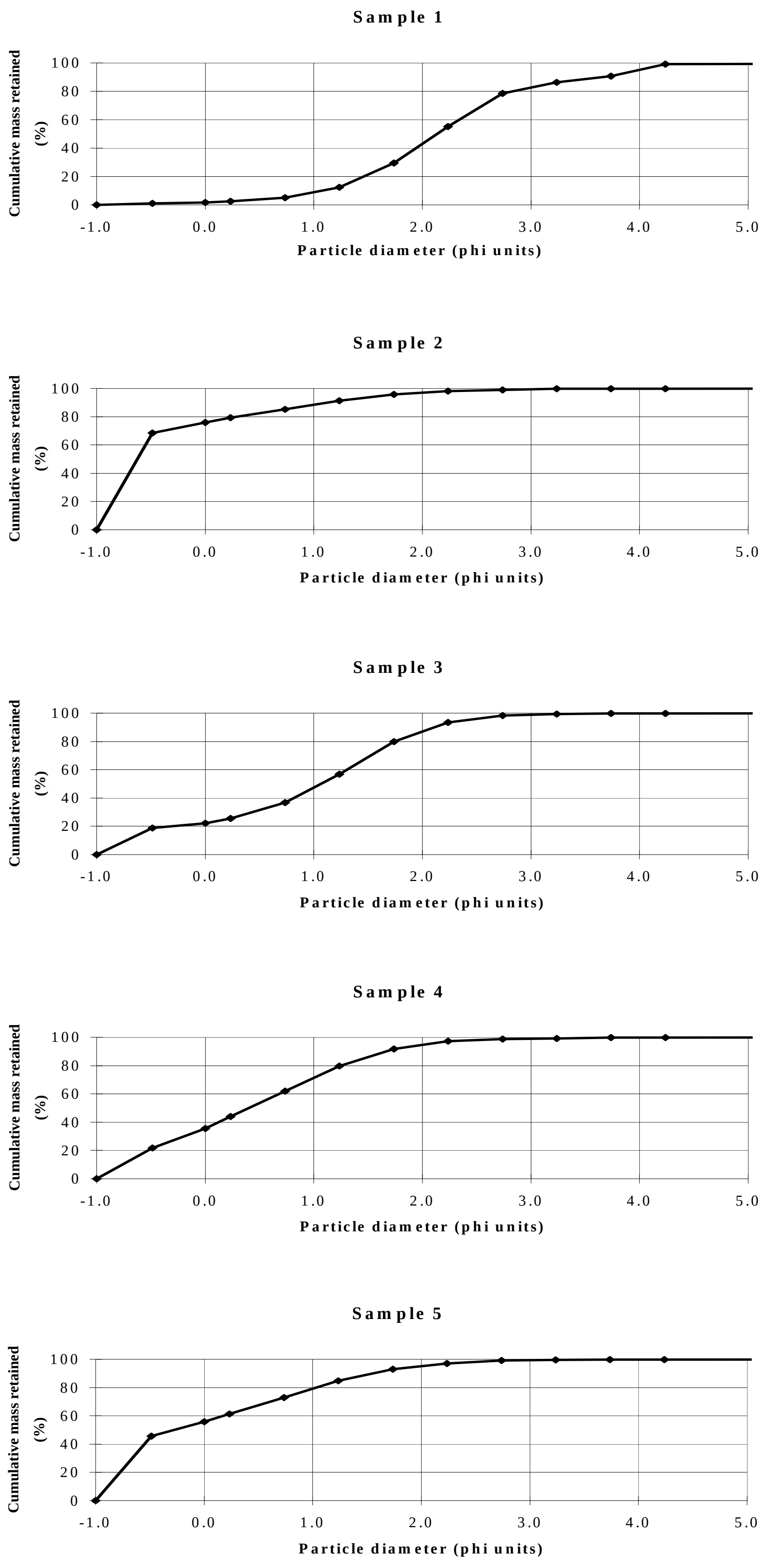


Figure 12.9b Grain size cumulative curves for samples 1-5 at cross-section 1.3, Site 1.

Table 12.14 Formulas for calculating mean and sorting values.

Mean	$Me_{ø} = 1/3(Ø_{16} + Ø_{50} + Ø_{84})$
Sorting	$S_{ø} = 1/4(Ø_{84} - ø_{16}) + 1/6.6(Ø_{95} - ø_{5})$

Note: $Ø_n$ is the phi-value corresponding to the n th percentile read off the from the cumulative curve.

The mean and sorting values for the five quadrants and five samples taken within the quadrants along transect 1.3, are shown in Table 12.15.

Table 12.15 Mean and sorting values for the five quadrants and five samples.

	Quadrants					**Samples**				
	1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Mean (ø-value)	-4.505	-5.127	-5.248	-4.894	-4.894	2.187	-0.290	0.797	0.400	0.036
Sorting	1.107	1.465	1.136	1.091	1.207	0.932	0.775	1.105	0.951	0.950

12.5.4 Macro-invertebrates

The results in this section represent two sampling sessions in May 2001 and February 2002, and are not representative of the full annual suite of invertebrate families occurring in the river. However, summer sampling does indicate the worst expected water quality, as there is no dilution effect from rain. There was a large difference between the types of animals found on large stones and ones found in sand. Increased species diversity was normally associated with complex substrata of cobble, gravels and sand. Riffles generally provided the best variety of aquatic habitats since they consisted of a mixed substratum (Gordon *et al.* (1992*)*. The samples from May 2001 (Table 12.16) had higher abundance scores and a noticeably higher number of taxa per sample than those from February 2002 (Table 12.17). There were however, two exceptions. In samples 1A4-01 and 1B4–02 the total number of families were equal while in samples 1A6 - 01 and 1B6 - 02 the number of families were higher in 2002. A total abundance score of 102 occurred within the sampled habitats in 2001 compared to 90 in 2002.

Table 12.16 **SASS-type scores for invertebrate samples collected on 16 May 2001**. Sample codes: 1A1-01, represents the following: 1 = site #, A = 16 May 2001, 1 = sample # and 01 = year. Substratum categories as in Table 12.3 with flow type classes shown in Table 12.6.

Class or Order	Family							
		Sample code	1A1-01	1A2-01	1A3-01	1A4-01	1A5-01	1A6-01
		Area (m)	1.3 x 0.8	1.3 x 0.3	1.0 x 0.4	1.4 x 0.5	1.2 x 0.3	2.0 x 0.5
		Habitat	BPF, MV+F	MRF, C+F	BPF, OL+ C+ B+G+F	SMF, OL+C+ G+B+F	FTF, C+ B+G	FTF, MV +OL
Amphibia	(tadpoles)		1	0	0	0	0	0
Osteichthyes	(fish)		1	0	0	0	0	0
Turbellaria	Planariidae-planaria		0	0	0	0	0	0
Annelida (Oligochaeta)	Lumbriculiidae		0	3	2	3	3	0
Decapoda	Brachyura (crabs)		1	0	1	1	0	0
Ephemeroptera	Baetidae		2	4	3	4	4	4
(mayflies)	Caenidae		0	0	2	0	0	0
Odonata	Coenagrionidae		4	0	0	2	0	3
(dragonflies,	Gomphidae		2	0	1	1	0	0
damselflies)	Libellulidae		0	2	0	0	0	0
Hemiptera	Pleidae		0	0	0	0	0	0
(bugs)	Notonectidae		0	0	0	0	0	0
	Veliidae		1	0	0	0	0	0
	Corixidae		3	0	0	2	0	0
Coleoptera (beetles)	Dytiscidae		0	0	0	0	0	0
Diptera	Simuliidae		0	4	2	0	4	4
	Chironomidae		3	3	2	2	3	4
Gastopada (snails, limpets)	Ancylidae		0	3	3	2	3	0
Total # of families		0	9	6	8	8	5	4
Abundance ratings		0	18	19	16	17	17	15
Total abundance rating score				102				

Table 12.17 SASS-type scores for invertebrate samples collected on 27 February 2002. Sample codes: 1B1-02, represents the following: 1 = site #, B = 27 February 2002, 1 = sample # and 02 = year. Substratum categories as in Table 12.3 with flow type classes shown in Table 12.6.

Class or Order	Family	Sample code	1B1-02	1B2-02	1B3-02	1B4-02	1B5-02	1B6-02
		Area (m)	2.0 x 0.2	1.0 x 0.5	1.5 x 5.0	1.5 x 0.5	1.5 x 1.0	2.0 x 0.2
		Habitat	BPF, MV + F	MRF, C+F	BPF, OL+ C+ B+G+F	SMF, OL+ C+G+B+ F	FTF, C+B+G	FTF, MV+OL
Amphibia	(tadpoles)		2	0	0	0	0	4
Osteichthyes	(fish)		0	0	0	0	0	0
Turbellaria	Planariidae-planaria		0	0	0	2	0	0
Annelida (Oligochaeta)	Lumbriculiidae		0	0	0	1	0	0
Decapoda	Brachyura (crabs)		1	0	0	0	0	1
Ephemeroptera	Baetidae		2	5	3	3	2	4
(mayflies)	Caenidae		1	3	3	2	0	0
Odonata	Coenagrionida		2	3	2	1	0	2
(dragonflies,	Gomphidae		2	0	0	2	0	0
damselflies)	Libellulidae		0	0	0	0	0	0
Hemiptera	Pleidae		4	0	5	0	0	0
(bugs)	Notonectidae		0	0	2	0	0	0
	Veliidae		2	0	0	2	0	2
	Corixidae		0	0	0	0	0	3
Coleoptera (beetles)	Dytiscidae		0	0	0	0	0	1
Diptera	Simuliidae		0	4	0	0	4	4
	Chironomidae		2	0	0	0	0	2
Gastropoda (snails, limpets)	Ancylidae		0	2	0	2	0	0
Total # of families		0	8	5	5	8	2	9
Abundance ratings		0	18	17	15	15	6	19
Total abundance rating score					90			

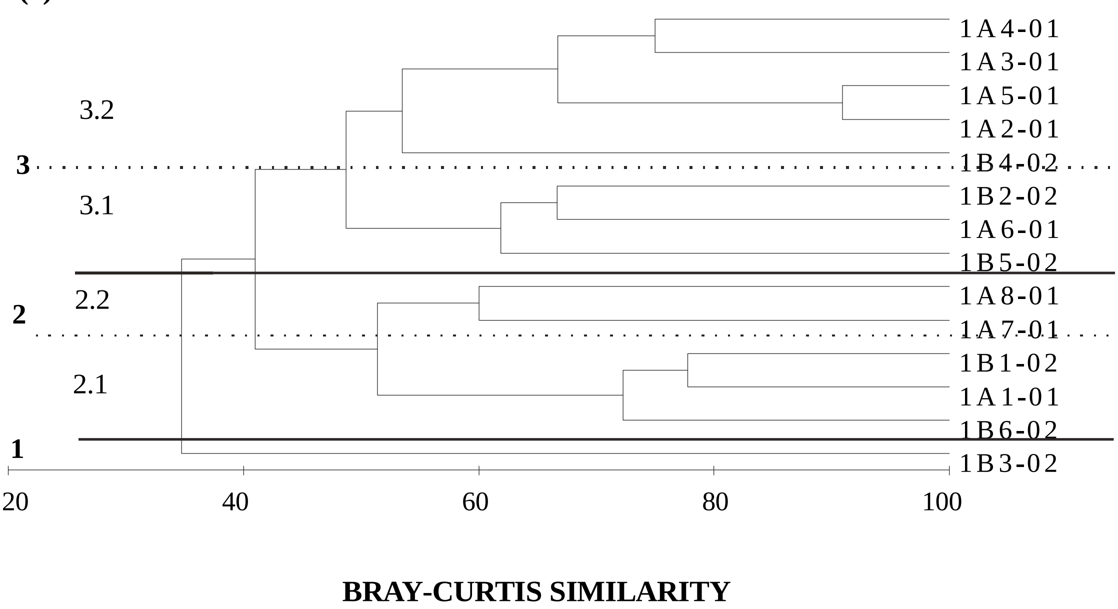
Much the same groups of invertebrate families occurred in both years. Although the study employed a modified version of SASS and the results were not actual SASS results, bioassessment results in this study can be used as a guideline of the general riverine ecosystem condition. Guidelines on SASS methodology and interpretation were reported in Thirion *et al.* (1995). The results shown in Tables 12.16 and 12.17 indicated that the Lourens River at Site 1 was in a fair to good condition in both years. The abundance of organisms gave a good indication of the ecological condition of the river. According to Thirion *et al.* (1995) high numbers of annelids and chironomids were normally an indication of organic pollution. The observed counts for the Lourens River indicated that although abundance scores for chironomids were higher in 2001 (averaging 2.5 and 0.67 respectively) than in 2002, they did not reach a score of 5 (>100 organisms per sample). The riverine condition at Site 1 most likely reflected the agricultural activities of the upper catchment (e.g. piggery and nursery).

The data from 2001 and 2002 were transformed into similarity clusters by using the presence / absence standardized transformation (Figure 12.10). The stress value for the MDS was 0.16, which means the ordination analyses results were reliable. In Figure 12.10a, group one was separated from groups two and three at 35% similarity level. Groups two and three were separated at 45% level. The first group consisted only of sample 1B3-02, invertebrates living in a mixed cobble+boulder+gravel+fine substratum overlain with organic litter and including Baetidae, Caenidae, Coenagrionidae, Pleidae and Notonectidae.

The second group was further subdivided into two subgroups, 2.1 and 2.2, at the 52% similarly value. Invertebrates found in subgroup 2.1 included Lumbriculiidae, Caenidae, Gomphidae, Pleidae, Corixidae and Dytiscidae. All the samples in this group had Decapoda (crabs), Baetidae, Coenagrionidae, Veliidae, Simuliidae and Chironomidae in common. The samples were all taken in marginal vegetation mixed with fines and organic litter and in flow types ranging from barely perceptible flow to fast turbulent flow. In subgroup 2.2 samples 1A7-01 and 1A8-01 formed a pair at 60% similarity. Both samples were taken in a mixed gravel-fines substratum covered with organic litter and had, Baetidae and Gomphidae in common. Invertebrates in samples 1A7-01 and 1A8-01 were taken from smooth and barely perceptible flow types respectively.

In addition, sample 1B2-02 having Caenidae and Ancylidae in addition and sample 1A6-01 having Chironomidae in addition. In subgroup 3.2 samples 1A2-01 and 1A5-01 formed a pair at the 92% similarity level while samples 1A3-01 and 1A4-01 formed a pair at 75% similarity level. Invertebrates in samples, 1A2-01 and 1A5-01 where taken from medium rippled and fast turbulent flow types respectively which had mixed underlying substratum consisting of cobble and sand (1A2-01) and cobble, boulder and gravel (1A5-01). Samples 1A3-01 and 1A4-01 were taken in a mixed substratum of cobble, gravel and fines covered with organic litter. This pair was characterised by slow flow types smooth (1A4-01) and barely perceptible flow (1A3-01). Sample 1B4-02 separated at 54% similarity level with the two pairs and was taken in mixed substrate types of organic litter on cobble, gravel, boulder and fines in smooth flow conditions. Subgroups 3.1 and 3.2 had invertebrate

(a)



(b)

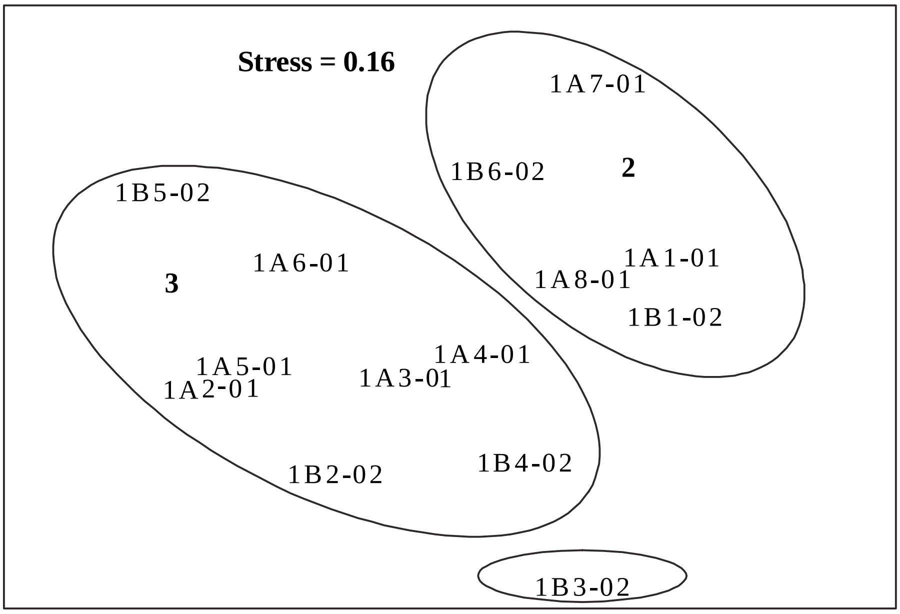


Figure 12.10 (a and b) Similarity analysis of invertebrate samples at Site 1 in 2001 and 2002 in (a) dendrogram and (b) MDS plot. Sample codes: 1B1-02, represents the following: 1 = site #, B = 27 February 2002, 1 = sample # and 02 = year. The solid line represents the spilt between the two communities.

families Lumbriculiidae and Baetidae in common. The invertebrate family Simuliidae was found in samples 1B4-02, 1A3-01 and 1A5-01 while Brachyura was found in samples 1A3-01 and 1A4-01. A complete list of samples with their community compositions can be seen in Tables 12.16 and 12.17.

Group three was further subdivided at 50% similarity into subgroups 3.1 and 3.2. All samples in subgroup 3.1 were taken from fast flow type groups. Samples in subgroup 3.1 had invertebrate families Baetidae and Simuliidae in common with sample 1A6-01 and 1B3-02 having Coenagrionidae.

The cluster dendrogram (Figure 12.10a) and MDS plot (Figure 12.10b) revealed that the main separation between group two and group three was between "fast" and "slow" flow types. All the invertebrate families sampled in group two were in the slow group. Samples in group three were in the fast group with the exception of families taken in sample 1A4-01. There was no overwhelmingly strong pattern concerning the sampling period or time in 2001 and 2002 and the specific groupings in the overall pattern because samples taken in both years appeared in most of the groups and pairs. In other words habitat was a stronger “grouper” than year.

12.5.5 Vegetation pattern change and hydrogeomorphology

The results of the vegetation sampling exercise indicated that considerable changes in the species composition of plant groups had occurred along Transect 1.3 at Site 1. The species recorded in April 2001 and in February 2002 with growth form, origin, habitat and weed status are shown in Table 2.18. Cluster analysis and ordination were performed on the vegetation data.

Site 1: Transect 1.3 2001

The data were transformed using the presence/absence standardised transformation and analysed for sample similarity (Figure 12.11).

The vegetation for 2001 was subdivided into two groups on the cluster (Figure 12.11a) and MDS plots (Figure 12.11b). The stress value was 0.03. Several subgroups could also be identified. Group one separated into subgroup 1.1 and 1.2 at 28% similarity. Group two further subdivided at the 12% similarity value into subgroups 2.1 and 2.2. Table 12.19 and Appendix 12.1 and 12.2 revealed a clear difference in species composition between groups one and two for 2001. There were three species present in group one and 17 in group two for 2001. In both groups the vegetation consisted predominately of herbaceous ground-storey species. This is clearly an indication of a disturbed area because a variety of trees, shrubs and herbs normally colonise flood benches, channel banks, terraces and flood plains. Channel bars are normally dominated by herbaceous species (Rowntree 1991). The groups were dominated by weeds and

Table 12.18 Vegetation species and their characteristics along Transect 1.3 at Site 1 for 2001 and 2002.

No	Species name	Growth form	Ann/Perr	Origin	Habitat 2001	Habitat 2002	Weed
	Vegetation 2001						
1	*Acacia mearnsii*	T (SL)	P	AN	FB		Yes
2	*Scirpus species*	H	A/P	I	FB, L		No
3	*Conyza canadensis*	H	A	AN	FB		Yes
4	*Distephanus angolensis*	H	*	AN	FB		Yes
5	*Dittrichia graveolens*	H	A	AN	FB		Yes
6	*Tarchonanthus camphoratus*	T (SL)	P	I	L		No
7	*Brachiaria eruciformis*	H	P	I	L		No
8	*Cynodon dactylon*	H	P	AN	L		Yes
9	*Eustachys paspaloides*	H	P	AN	L, CB		Yes
	Vegetation 2002						
1	*Tamarix chinensis*	H	P	AN		RMB	Yes
2	*Fumaria officinalis*	H	A	AN		FB	Yes
3	*Conyza bonariensis*	H	A	AN I		FB	Yes
4	*Hypochaeris radicata*	H	P/A	AN		FB	Yes
5	*Lolium perenne*	H	P	AN		FB	Yes
6	*Oxalis pes-caprae*	H	P	I		FB	Yes
7	*Rumex acetosella*	H	P	AN		FB	Yes
8	*Lactuca serriola*	H	P	AN		FB	Yes
9	*Verbena officinalis*	H	*	AN		L	Yes
10	*Brachiaria serrata*	H	A	I		L	Yes
11	*Cyperus esculentus*	H	P	AN		L	Yes
12	*Polypogon* species	H	P/A	AN/I		L	Yes
13	*Panicum gilvum*	H	*	AN		L	Yes
14	*Rumex* species	H	P	AN		L	Yes
15	*Bidens pilosa*	H	A	AN		F	Yes
	Species in both 2001 and 2002						
1	*Pennisetum clandestinum*	H	P	AN	HT, RMB, FB, L	HT, RMB, FB, L, LB	Yes
2	*Tropaeolum majus*	H	A	AN	RMB	RMB	Yes
3	*Populus x canescens*	T	P	AN	RMB	RMB, FB	Yes
4	*Plantago lanceolata*	H	A	AN	FB, L	FB	Yes
5	*Convolvulus arvensis*	H	P	AN	L, LB, F	LB, F	Yes
6	*Commelina benghalensis*	H	P	AN	L, CB, LB	L, LB, F	Yes
7	*Cyperus longus*	H	P	I	L	L	No
8	*Polygonum* species	H or S	A/P	AN/I	L	L	Yes
9	*Ipomoea purpurea*	H	A	AN	LB, F	LB, F	Yes

Key to abbreviations

Growth Form: SL = Seedling; H = Herb; S = Shrub; T = Tree
Ann/Perr: A = Annual; P = Perennial
Origin: AN = Alien; I = Indigenous
Habitat: HT = High terrace; LB = Left bank; RMB = Right hand macro-channel bank; FB = Flood bench; L = Lateral bar (right hand active channel bank); CB = Channel bed; F = Floodplain
* Information not available

included both indigenous and alien plants. Both annual and perennial plant species were recorded (Table 12.18).

The colonisation of the different vegetation groups on the cross-sectional features (riverine geomorphic units) found at Transect 1.3 are shown in Figure 12.12 and Table 12.18. Species in group one were associated with the lateral bar, channel bed, left bank and flood plain. Species in group two were restricted to the high terrace, right hand macro-channel bank, flood bench, and lateral bar. Results suggested that some species may be site specific. For example *Tropaeolum majus* and *Populus x canescens* were restricted to the right hand macro-channel bank whereas *Polygonum* species, *Ipomoea purpurea* and *Cyperus longus* were located on the lateral bar. The results indicated that many species did not discriminate between two adjacent geomorphic features, but were largely excluded from a third. The exceptions were *Pennisetum clandestinum and Commelina benghalensis.* The flood bench and lateral bar surfaces supported many species that were not found on other morphological units. Both groups consisted of a combination of wet and dry bank vegetation. Species common to groups one and two are indicated in Table 12.19.

The results indicated that the split between groups one and two was, with a few exceptions, mainly the result of different habitat preferences. Group one was situated on the right bank whereas group two was on the left bank (Figure 12.12 and Table 12.18).

Site 1 Transect 1.3 2002

The data for 2002 were transformed in the same way as for 2001 using the presence/absence-standardised transformation. Three groups were identified. The resulting dendrogram and MDS plots from the Primer analysis (Figure 12.13a and b) revealed that the main separation between groups one and two was at 2% similarity level. Group three separated at 1% similarity level from group two and three. This group differed completely from the other two groups. Within each main group there were several subgroups that could be identified. Group one separated at 10% similarity into subgroups 1.1 and 1.2. Group two was further subdivided into subgroups 2.1 and 2.2, at the 12% similarity value. A summary of the species composition for each group is given in Table 12.19. There were nine species present in group one, 18 in group two and one in group three. In all three groups the vegetation consisted predominately of herbaceous ground-storey species, mostly weeds including both indigenous and mostly alien plants. Both annual and perennial plant species were identified (Table 12.18).

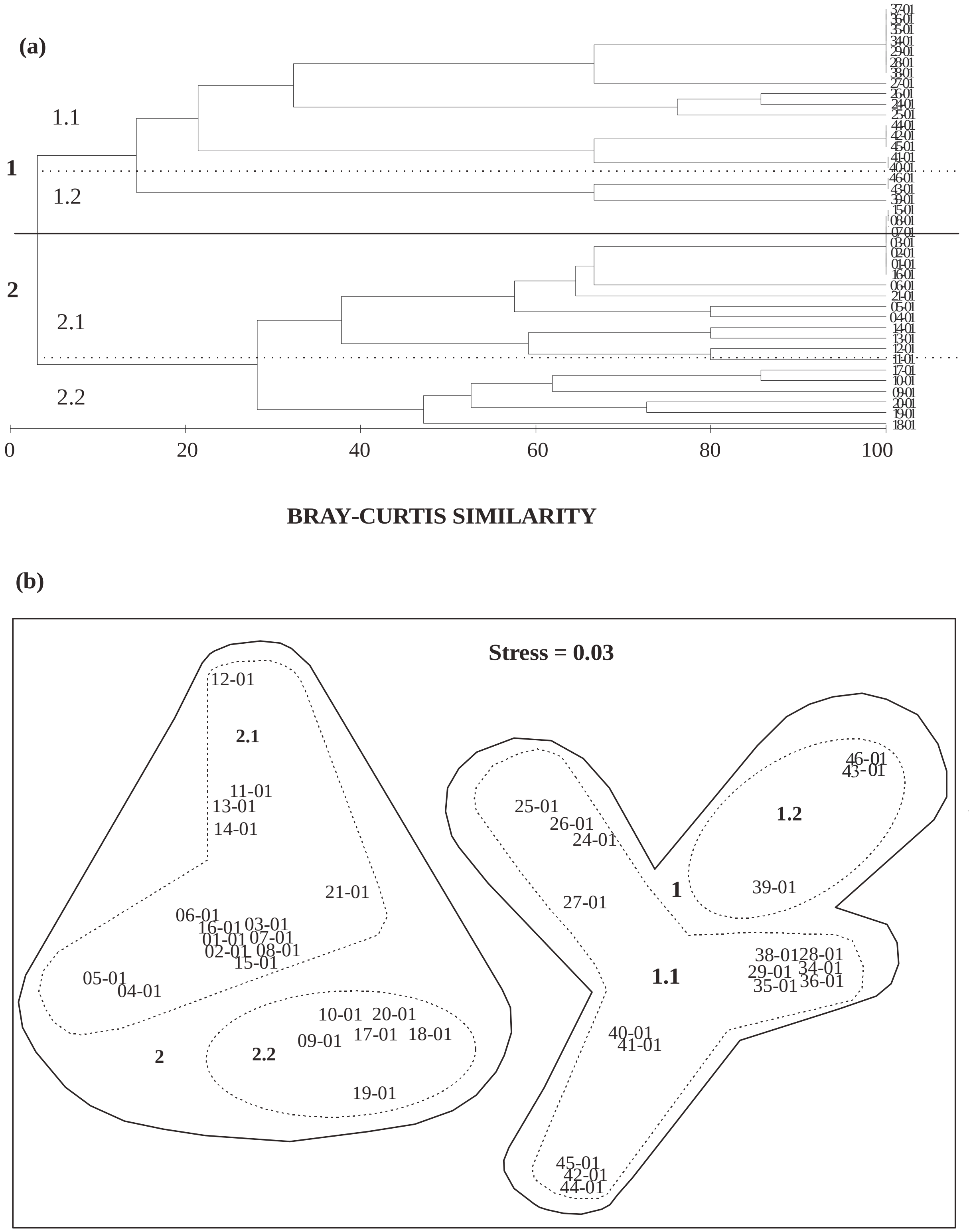


Figure 12.11 (a and b) **Identification of vegetation groups at Site 1, Transect 1.3, 2001**. Vegetation data codes: 01-01, denotes the following: 01 = quadrant #, 01 = year. The solid line represents the spilt between the two groups.

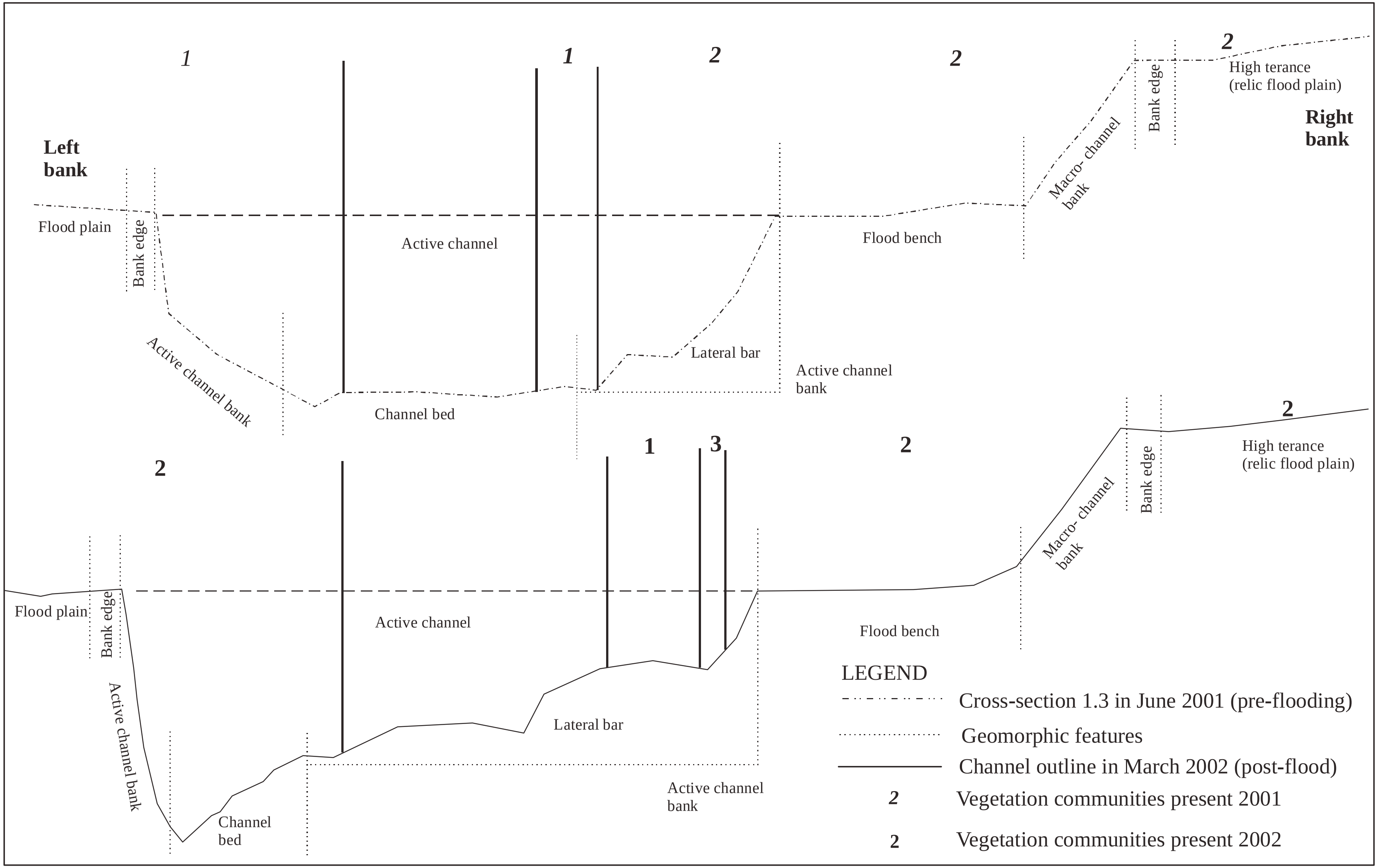


Figure 12.12 Geomorphological features and associated vegetation groups present at Site 1, Transect 1.3, in 2001 and 2002 (adapted from Rowntree 1991).

The geomorphological units and their associated plant groups are indicated in Figure 12.12 and Table 12.18. Species in groups one and three were entirely restricted to the lateral bar whereas species in group two were found on all the geomorphological units except the channel bed. Results suggested that species in group one and three were position specific. As was the case with 2001, the results indicated that many species did not discriminate between two adjacent geomorphic features, but were largely excluded from a third, with the exception again being *Pennisetum clandestinum.* The flood bench and lateral bar surfaces again supported many species that were not found on other morphological units. Both group one and three consisted of wet bank vegetation. Species in group two consisted of a combination of wet and dry bank vegetation.

Species common to group one and two and two and three are indicated in Table 12.19. The cluster and MDS plots revealed that the overriding influence on the distribution pattern of species in group one was the frequency of inundation and the susceptibility of the plants to destruction. Group one occurred only on the lateral bar and river margin under preferred wet conditions. Group two did not show any clear surface preferences.

Table 12.19 Plant groups for Transect 1.3, the Lourens River (see also Table 12.18 and Figures 12.11 & 12.13).

2001	**2002**	
Group 1	**Group 1**	**Group 3**
Ipomoea purpurea	*Brachiaria serrata*	*Verbena officinalis*
	Cyperus esculentus	
Group 2	*Polypogon species*	**Species common to both groups 1 and 2**
Pennisetum clandestinum *	*Panicum gilvum*	*Commelina benghalensis* * +
Populus x canescens *	*Rumex species*	
Tropaeolum majus *	*Cyperus longus*	**Species common to both groups 2 and 3**
Acacia mearnsii	*Polygonum* species	*Verbena officinalis*
Scirpus species		
Plantago lanceolata *	**Group 2**	
Conyza canadensis	*Pennisetum clandestinum* *	
Distephanus angolensis	*Populus x canescens* *	
Dittrichia graveolens	*Convolvulus arvensis* *	
Tarchonanthus camphoratus	*Tamarix chinensis*	
Eustachys paspaloides	*Fumaria officinalis*	
Brachiaria eruciformis	*Conyza bonariensis*	
Cynodon dactylon	*Hypochaeris radicata*	
Polygonum species	*Lolium perenne*	
Cyperus longus	*Oxalis pes-caprae*	
	Rumex acetosella	
Species common to both groups	*Lactuca serriola*	
	Bidens pilosa	
Convolvulus arvensis *	*Ipomoea purpurea*	
Commelina benghalensis * +	*Plantago lanceolata* *	
	Tropaeolum majus *	

+ Persistent species in group 1 for 2001 and 2002.
* Persistent species in group 2 for 2001 and 2002**.**

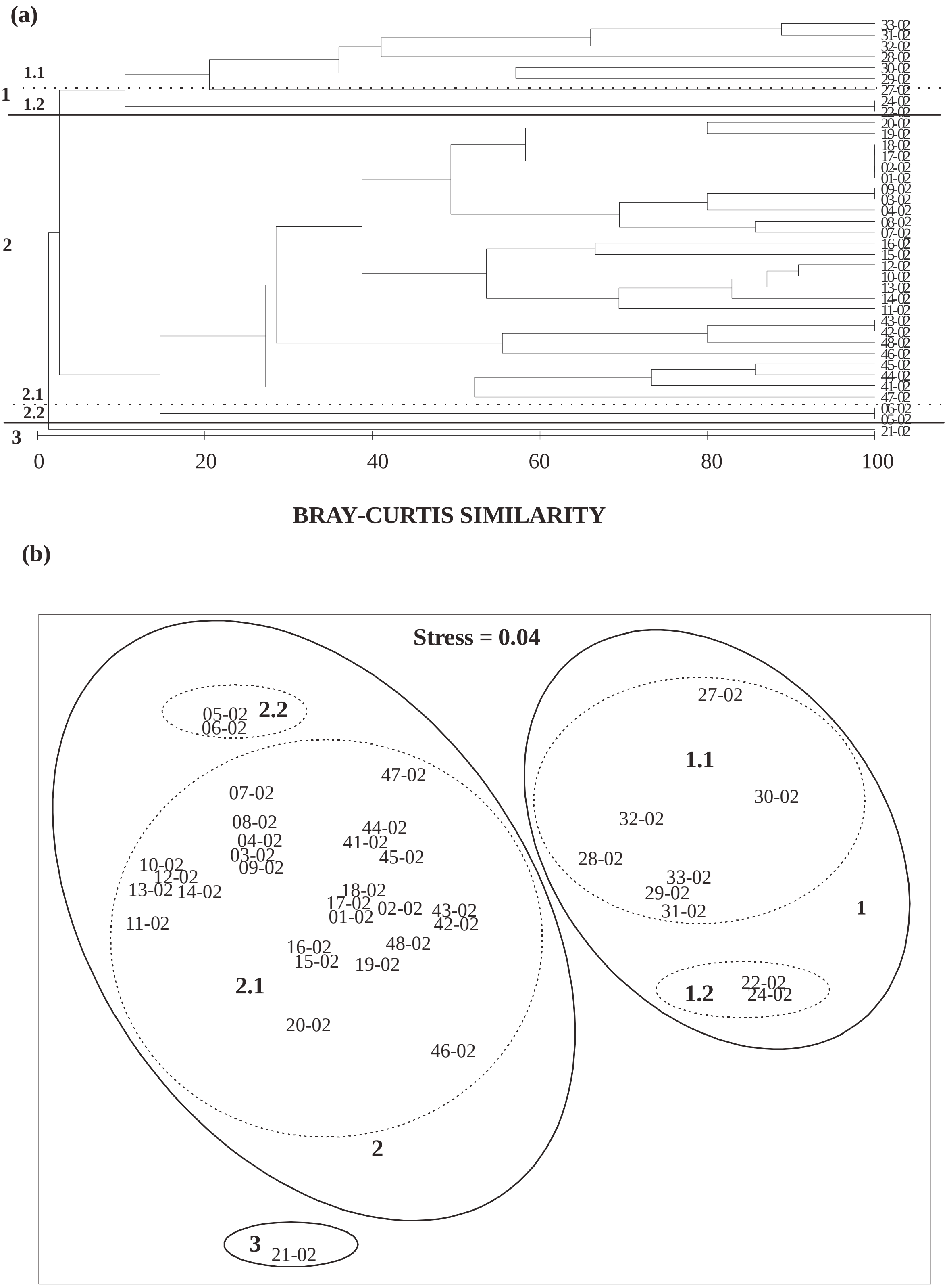


Figure 12.13 (a and b) **Identification of vegetation groups at Site 1, Transect 1.3, 2002**. Vegetation data codes: 01-02, denotes the following: 01 = quadrant #, 02 = year. The solid line represents the spilt between the two groups.

12.6 Discussion

This discussion is divided into three sections. The first one addresses changes in channel morphology, the second macro invertebrate fauna and the last one, vegetation change and hydrogeomorphology.

12.6.1 Channel morphology

River flow varies with time and river channels continually change position and shape because of hydraulic forces acting on bed and banks. These changes mostly take place over a long period of time. Catchment and site variables are the two sets of variables determining channel form and process. The former determines the runoff and sediment regime of the river while the latter control the stability of the channel (Hughes 1977; Dollar & Rowntree 1995; Gurnell 1997; Wohl 2000). The geometry and stability of the channel is therefore determined by the interaction between flowing water, the magnitude and characteristics of sediment load, the riparian vegetation and the bed and bank characteristics.

According to Gordon *et al.* (1992) substrate normally refers to the particles on the riverbed, both organic and inorganic. Generally, coarse materials (gravel, boulders) are associated with high-energy environments and finer materials (sand, silt) with low energy environments. The instability of the riverbed also decreases as the particle size decreases, although this will also depend on the mix of particle sizes and shapes. A wide range of particle sizes generally characterises the bed material of gravel-bed rivers. Bed load varies in size from fine sand to coarse cobbles. Many studies have focused on the role of this variation in controlling channel morphology, longitudinal form and the occurrence of aquatic organisms (Gordon *et al.* 1992). Riverbed materials are also subjected to dislodgment or removal during flood events and human activities such as dredging. The composition of bed material may be altered by sediment influxes from upstream or upland erosion and by channel modification. Generally these changes tend to influence downstream river conditions (Gordon e*t al*. 1992).

Channel form adjustments take place through changes in cross-section (width and depth), channel gradient and channel plan. The mode of adjustment is controlled by the availability of energy, which can be related to stream power and resistance. Adjustment processes that effect entire fluvial systems often include upstream-progressing degradation, downstream aggradation, channel widening or narrowing and changes in the quantity and characteristics of its sediment load. Both aggradation and degradation represent systematic changes in bed elevation over time (Hughes 1977; Dollar & Rowntree 1995; Gurnell 1997; Wohl 2000).

Evidence for instability in the Lourens River, Site 1, can be seen in the form of bed and bank erosion, in-channel deposition, bar formation and channel shifting or migration. Variables that played a role in the observed changes in channel morphology discussed in Sections 12.5.1 and 12.5.2 included: vegetation composition and cover, channel bed and bank characteristics and changes in the rainfall.

The sediment results indicated that the left bank is a non-cohesive sandy bank with very small fractions of silt and clay and a covering of herbaceous weeds.

According to Thorne (1982) failure of non-cohesive banks by flow action occurs through the dislodgement of individual grains from the bank surface. Bank retreat results when the flow scours the bed at the base of the channel bank (increasing bank angle and length) to bring about gravitational failure of the intact bank. Processes of weakening (prolonged rainfall events) act on the intact bank material to reduce strength and decrease bank stability. The saturated bank will be very unstable because of the decrease in strength and the increase in unit weight (Thorne 1990). The flood of 2001 resulted in increased current speeds. These exerted greater shear stresses on the river bed and banks and resulted in the gravitational failure of the left bank as described in Section 12.5.1. Failure mechanisms and bank erosion processes act in different ways to produce bank retreat and are significantly affected by vegetation. Vegetation reduces bank erosion by slowing flow velocity, trapping sediments among stems and binding soil, making it resistant to the erosive forces of water.

Vegetation type greatly influences bank protection and stability (Thorne 1982; 1990). Grasses, shrubs and herbaceous vegetation have low biomass and are shallow rooted. They tend to provide a good surface cover and a dense near-surface root mat that does not affect stability with respect to deep bank failures (Rowntree 1991). Vegetation also influences channel form through its combined affect on geomorphic processes. Woody vegetation has a more varied impact on channel processes through the development of debris dams, increased bank stability and increased boundary roughness. Herbaceous vegetation reduces scour and enhances sediment deposition. Generally, grassy banks are associated with wider shallower channels and tree lined banks with narrow deeper channels (Rowntree 1991).

The bank erosion at Site 1 was thus the result of increased velocities scouring the sandy and dry non-cohesive left bank which was covered by shallow rooted herbaceous vegetation. In-channel deposition and bar formation were also the result of the flood events. The gabion mattresses located upstream from the site, on the right bank and at the beginning of the site, on the left bank, probably played a role in the results obtained. When gabion mattresses are used in bank stabilization on one side of the river they should be located on the other side as well (K. Rowntree, Geography Department, Rhodes University, pers. comm.). According to Wohl (2000), deposition occurs where sediment supply overwhelms flow transport capacity. Bed fill occurred during a moderate flood in August 2001, with sediment supplied during the flood by observed upstream bank erosion and failure. Infilling of 0.5 m also occurred at the lateral bar, on the right bank. This localized deposition resulted from a change in channel geometry such as a decrease in channel-bed gradient or an increase in channel width which caused flow velocity to decrease (Wohl 2000).

Likely erosion points in a river would therefore probably be located on the outside bends of the channel. Indicators of this could be: steep banks covered with sparse or alien vegetation, non-cohesive slopes or sediments, channel modifications upstream or at the site and deep narrow channels.

12.6.2 Macro-invertebrate fauna

The cluster (Figure 12.10a) and MDS plots (Figure 12.10b) revealed that the main split between the communities in Site 1was between "fast" and "slow" flow types. All the invertebrate families sampled in community two were in the slow group with the exception of families in sample 1B6-02. Samples in community three were in the fast group with the exception of families taken in sample1A4-01. All three communities had crabs (Decapoda), mayflies (Caenidae), damselflies (Coenogrionidae), bugs (Veliidae), dragonflies (Gomphidae) and non-biting midges (Chironomidae) in common. Communities two and three had tadpoles (Amphibia), segmented worms (Lumbriculiidae), mayflies (Baetidae) and blackflies (Simuliidae) in common. Community one had bugs (Pleidae) in addition whereas community two had dragonflies (Libellulidae) in addition. Community three had bugs (Corixidae) and beetles (Dytiscidae) in addition.

There was not an overwhelmingly strong pattern or change in community structure for the samples between 2001 and 2002 concerning the sampling period or time (during winter and summer) because samples taken in both years appeared in most of the groups and pairs. According to the SASS-type scores the results indicated that 2001 samples had higher abundance scores than 2002 samples and also a higher number of taxa per sample. The general health of the river was fair to good in 2001 and 2002 respectively. Generally higher abundance values were found in habitats containing vegetation or organic litter accompanied by barely flowing water in both years.

The absence of insects such as Trichoptera, Plecoptera, which formed the basis of the natural fauna of the foothill zone (Tharme e*t al.* 1997), indicates a considerable loss of biotic integrity at Site 1. The presence or absence of other taxa can also be used to highlight changes in riverine conditions. High numbers of midges and black flies are frequently associated with poor water quality. The results (Table 12.16) indicated that abundance scores for midges and black flies in 2001 were three and four respectively whereas in 2002 (Table 12.17) it was four in both cases. Chironomid numbers increase in nutrient-rich water while increases in the suspended organic particles in the water column increase the number of simuliids (Tharne e*t al.* 1997). The unusually high rainfall and upstream agricultural activities might explain the fact that there were fewer species in 2002 than in the winter of 2001, with the exception being the above mentioned chironomid and simuliid numbers. Flood waters scoured the bed and banks and increased the amount of suspended sediments in the water column. The animals probably were either swept away or retreated deeper into the riverbed (King 1983). There seems to have been a deterioration in habitat or water quality in the Lourens River at Site 1 because the river no longer supported the complement of invertebrates reported by Ractliffe (1991).

Overall, substratum type was a consideration for animal samples taken in vegetation with fines, gravel had more taxa present and the invertebrate families grouped into communities of fast and slow flow types. As indicated by Rowntree & Wadeson (1999) there was a strong link between hydraulic biotopes, flow type and substratum characteristics. Habitat availability might also play a role in the absence or presence of some taxa. Vegetation results of 2002 indicated that no vegetation was present within the channel. This fact might have reduced the number of species because some species prefer to live in aquatic or marginal vegetation.

12.6.3 Vegetation pattern change and hydrogeomorphology

The results presented in Section 12.5.5 indicated that the groups of 2001 differed from those of 2002 mainly with respect to species composition and distributional pattern (Table 12.19 and Figure 12.12). Nine species that were present in 2001 (Table 12.18), located on the flood bench, 1ateral bar and channel bed, disappeared in 2002 and were replaced with fifteen new species. These new species mostly colonised the flood bench and the lateral bar. However, there were also nine species that were common to 2001 and 2002. These persistent plant species were primarily associated with more than two geomorphological units and occurred particularly on areas that were not so regularly inundated such as the right hand macro-channel bank, left bank and high terrace.

The results indicated that herbaceous species dominate Site 1. These species were mostly exotic weeds (Table 12.18). Only remnants of tolerant indigenous flora remained. Exotic species have been introduced from elsewhere in the world, become naturalised and many have become abundant, even to pest proportions (Bromilow 1995, Tickner *et al*. 2001). Their ability to disperse over short distances, rapid seed growth and establishment, wide tolerance of variation in the environment, self compatibility and strong competitive ability make exotic species well adapted to invade riparian zones (Hill 1977; Tickner 2001). Historically the distribution of aliens along the Lourens River was extensive, although it decreased somewhat with urbanisation (Tharme *et al.* 1997). Escaped exotic species from gardens also then established themselves as dominants at Site 1.

The observed plant distributions and change in group composition in this study were at least in part controlled by a complex mix of inundation frequency, the susceptibility of plant species to damage by flooding events experienced in 2001, and to seed dispersal, moisture and nutrient availability. Most of the species located on the flood bench, lateral bar and left and right channel margins were removed or buried by the flood events of 2001. Bendix & Hupp (2000) reported that although floods played a major role in the establishment and survival of plant species, non-hydrogeomorphological environmental factors such as temperature and precipitation were also important. According to Hupp (2000) the following two parameters are needed in order for a given species to grow vigorously on a particular geomorphic feature: a suitable site for germination and establishment, and the ambient environmental conditions that would allow persistence at least until reproductive age. The distributional patterns of vegetation in riverine systems across geomorphological units may be mainly driven by the tolerance of plant species to specific geomorphological processes at the severe end of a stress-equilibrium gradient and by competition with other species at the other end (Hupp 2000). Floods depositing sediments and destroying pre-existing vegetation create colonisation sites. The occurrence or the lack of floods subsequent to germination may influence seedling survivals. Floods may also play an important role in dispersing propagules to colonisation sites (Bendix & Hupp 2000).

Flood disturbance thus resulted in vacant habitat niches ready for colonisation by herbaceous exotics. A new habitat for colonisation was created on the right hand active channel bank in the form of a lateral bar. Light and space were thus easily available and competition from previously existing species was greatly reduced. Continued wet conditions after the flooding events proved ideal for the establishment of invasive species both by reproduction of seeds and by vegetative spread. However, it is important to note that some species may be common where floods are severe not because they can survive floods, but because they are rapid colonisers of flood-cleared surfaces (Bendix & Hupp 2000). According to Tickner e*t al*. (2001) cycles of erosional and depositional processes can create new habitats but can also make establishment difficult. It is thus the hydrogeomorphic processes operating differently on different morphological units that actually affect the plant patterns, not the units *per se* (Hupp 1990). It is concluded that vegetation pattern change at Site 1 was mainly the result of flood-related factors.

2.7 Conclusion

The impacts of the 2001 flooding events can clearly be seen in this study. The results obtained are similar to those reported by a number of authors. The following conclusions can be made.

- Channel morphology was influenced by flow changes. Channel deepening, channel migration, in-channel deposition, bank erosion and bar formation were the result of increased velocities.
- Hydraulic biotope classes changed in response to high rainfall and resulting higher discharges.
- The substratum and discharge played a major role in the flow characteristics of the different hydraulic biotopes.
- The riverine ecosystem was in fair to good condition but was somewhat degraded as shown by low diversity and abundances of invertebrates.
- Floods increased the subsequent diversity of plant species.
- The distribution of different sized sediments was correlated with channel processes such as degradation and aggradation, channel morphology and the occurrence of different assemblages of aquatic organisms.

13 THE KUILS RIVER

Ruth-Mary Fisher and Theo Scheepers, Department of Earth Science, University of the Western Cape

Not all the data collected are in this report. The full results are in the M.Sc. thesis of Miss Ruth-Mary Fisher entitled: Fisher, R.C. (in prep.). The impacts of channelisation on the geomorphology and ecology of the Kuils River, Western Cape, South Africa. Unpublished M.Sc. thesis. University of the Western Cape. Bellville.

13.1 Overall objective, background to the study and work plan

The overall objective of the project was to study a river where active management was taking place and then to record the river's response. The Kuils River was chosen as a study river because it is an urban river that was actively being channelised. Land-use in its catchment is largely urban and industrial. Land-use changes, especially urbanisation and waste-water inputs, have changed the flow of the river to perennial. Increased storm-water run-off has heightened the risk of flooding in the lower parts of the river. From January to December 2000, Ninham Shand Consulting Engineers were contracted by the City of Cape Town to "upgrade" the channel from the Van Riebeeck Road Bridge past Pioneer Street Bridge up to the gabion weir at Site 2 (Figure 13.1 & 13.2), in order to reduce the flood risk in the Kuilsrivier central business district. The term "upgrading" refers to increasing the carrying capacity of the channel by making it wider and deeper (M. Luger, Ninham Shand, pers. comm. 2002). The literature review reveals that the term channelisation describes the same activities; this term will therefore be used to describe what was done to the Kuils' channel.

The Kuils River offered the opportunity to study the effects of channelisation on the geomorphology (substratum and flow types that provide habitats) and ecology (macro-invertebrate and vegetation assemblages) of the river. The objective was to initially record in detail the geomorphological and ecological characteristics of the river, at reaches upstream, within and downstream of an area of channelisation, and to then track changes over one year of engineering activities and winter floods.

Data collection started early in the 2000/2001 summer, and was completed before the autumn rains started. There was extensive learning of new techniques in this period, which guided the second summer of data collection. The sites were observed during the winter floods of 2001, and features such as channel geometry re-measured when changes occurred. The full sampling programme was repeated in the second summer (2001/2002), in order to assess short-term change.

13.2 Study area

The Kuils River rises at a low altitude in the eastern foothills of the Tygerberg Hills in the Durbanville area. It was a small, sluggish river, which historically dried up each summer. Presently its runoff originates mainly in the storm water system of the Durbanville residential areas and emerges into a concrete-lined canal near Kanonkop (Ninham Shand 1999). From the source to the confluence with the Eerste River its catchment size is 261 km^2. The river is in a low-lying area with only the upper foothill, lower foothill and lowland geomorphological zones (Figure 13.1).

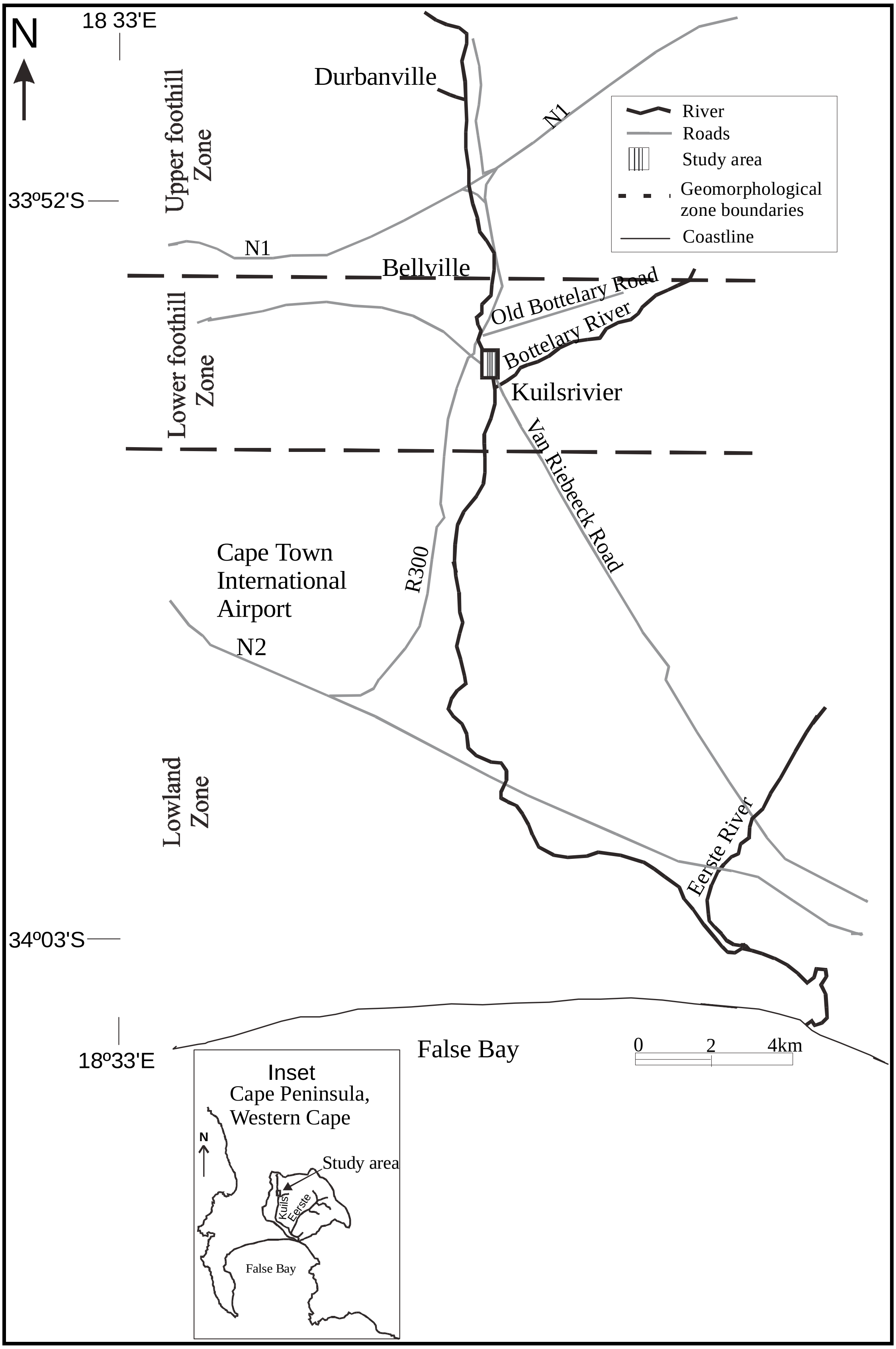


Figure 13.1 **Kuils River and catchment within the Cape Peninsula, Western Cape showing major suburbs, roads, geomorphological zones and study area.** (After Ninham Shand & Chittenden Nicks 1999)

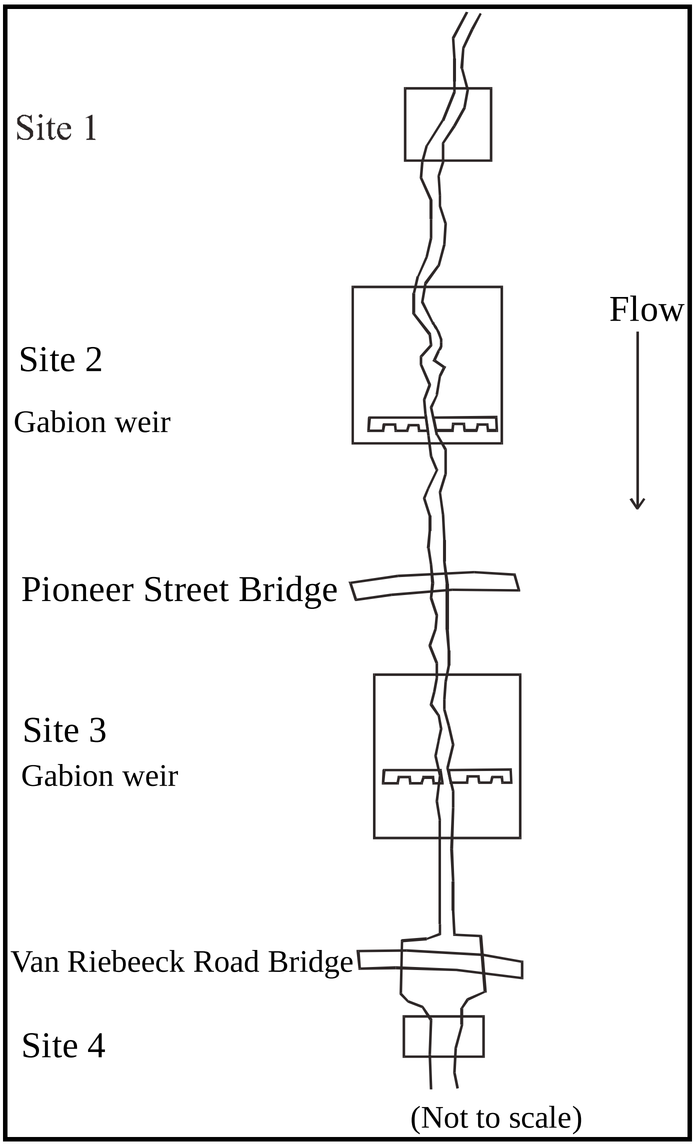


Figure 13.2 Location of the four sites within the study area (December 2000).

Its catchment is in the winter rainfall region of the Western Cape. An annual average rainfall of 555mm was measured at Durbanville for the period 1969-2000 (Figure 13.3).

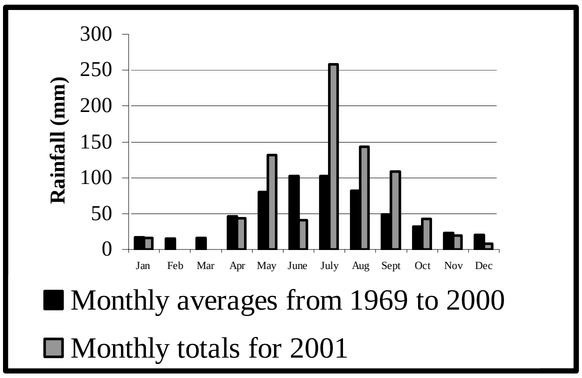


Figure 13.3 Rainfall data measured in Durbanville. (Unofficial record, T. de Villiers, local farmer, Durbanville area)

Catchment geology consists of rocks of the Malmesbury Group, which underlie the riverbanks and surrounding slopes in the upper reaches. The Malmesbury Group consists of quartzites, phyllites, greywacke and shales of Pre-Cambrian age. The rocks are covered with recent thin deposits of turf and loamy sands with some alluvial deposition (Grindley 1982; Shand *et al.* 1994; Ninham Shand & Chittenden Nicks 1999) down to the N1/R300 interchange. Clayey slopes occur at the interchange further away from the river and its floodplain. The river course flattens at the footslopes of the Tygerberg. The foothills of the Tygerberg phase into the Cape Flats coastal plain where alluvial soils are also clayey, but fine organic matter is present as well. The river then enters the coastal plain and passes through acidic sands with some local neutral to calcareous dune sands also present (Figure 13.1) (Ninham Shand & Chittenden Nicks 1999).

The natural renosterveld vegetation of the Tygerberg Hills has been replaced by agricultural land, urban development and alien vegetation (Grindley 1982; Ninham Shand 1999). Alien vegetation includes Port Jackson willow (*Acacia saligna*), Rooikrans (*Acacia cyclops*), Black wattle (*Acacia mearnsii*), White poplar (*Populus canescens*), *Sesbania* and other species (Ninham Shand 1999; Ninham Shand & Chittenden Nicks 1999).

Macro-invertebrate groups consist of hardy species in the river section between Van Riebeeck Road and the R300 (Figure 13.1). These include damselfly nymphs, diving beetles, dragonfly nymphs, midge larvae, snails and limpets. There are no indigenous fish, with only Carp, an exotic fish, known to inhabit the river (Ninham Shand 1999).

Four different reaches were identified within the study area. A representative study site was chosen within each reach. The four study sites (Figure 13.2) were:

- Site 1, upstream of all planned channelisation activities, acted as the control site. It had high, near vertical, eroding sand banks. The banks were covered with kikuyu, other grasses and a few alien trees. The site was approximately 46 m long with a narrow, confined, slightly meandering channel. Three cross-sections were established at the site in the summer of 2000/2001.
- Site 2, next downstream to Site 1 and originally similar to it, also showed signs of severe erosion with a confined macro-channel between high crumbling banks, an active channel meandering within this, and deep erosion dongas entering on either sides. It was approximately 104 m long, and due to be widened and the bank slopes reduced. It was located immediately upstream of the reach most recently channelised, which had been completed in December 2000. This site was chosen to record how ecological, geomorphological and hydraulic features would be affected as channelisation proceeded. The banks were covered with kikuyu, weeds and a few alien trees. Four cross-sections were established at the site in the summer of 2000/2001. After initial data collection at this site, it was widened in January and February 2001, and the banks re-graded to a milder slope and planted with kikuyu.
- Site 3 was in the already channelised stretch downstream of Site 2. It was 165 m long and fairly straight. It fell within the reach stretching from upstream of Van Riebeeck Road Bridge to downstream of Pioneer Street Bridge. It was widened and deepened in 2000, before the study began, and had a wide, shallow trapezoidal channel 0.5 m deep, with banks stabilised by kikuyu grass and a central earth-lined gully. The low-flow channel conveyed a discharge of $1.5 m^3 s^{-1}$, while the floodplain area could convey a discharge up to $140 m^3 s^{-1}$. Local bank reinforcement, in the form of thin rock mats, was introduced from the top of the slope down to

1m above the low water level. A gabion, designed to dissipate energy, was constructed within the section. Five cross-sections were established at this site in the summer of 2000/2001.

- Site 4 was in the most downstream reach, within the commercial district of Kuilsrivier. This section was canalised in 1993/1994. It is now a trapezoidal-shaped concrete channel, with a central gully. The concrete lining is 14 m wide in the invert and 20 m wide across the top. It is 1.175 m deep and has side slopes of 1:3. Two cross-sections were established at this site in the first summer of data collection.

Key questions posed in the study were as follows.

- *Geomorphology.* What is the cross-sectional area and shape of representative study sites in each reach? What substrata occur in each reach?
- *Macro-invertebrates.* What aquatic macro invertebrate families occur at each site? What do these indicate about water quality and the general health of the river? How might they be related to, or respond to, channelisation activities?
- *Vegetation.* What species of plants occur in the water and on the banks at each site, and how might they be related to, or respond to, channelisation activities?
- How does biodiversity differ at the sites?
- How do the sites respond to winter floods?

13.3 Methods

The sampling was done during the summer months (December to March 2000-2001 and 2001-2002) and repeated in winter (June to August 2001) if flood-induced ecosystem changes were observed (Table 13.1).

Table 13.1 Dates when the different parameters were sampled or measured.

Parameter measured/sampled	First summer sampling	Winter sampling	Second summer sampling
Cross sections	December 2000	July 2001 (site 1 and 2 only)	March 2002
Habitat maps	February to May 2001	July 2001 (all sites)	January to March 2002
Macro invertebrates	April to May 2001		March 2002
Vegetation	January to March 2001		January to February 2002
Discharge		May to July 2001	

13.3.1 Geomorphology: cross-sections and channel outlines

The cross-sections were surveyed using the tachometry method of optical distance measurement. A theodolite (Leica TC307 model) was used to obtain readings reflected off a Standard Leica prism fixed on a staff. The horizontal distance to the staff, the difference in surface elevations and the location in reference to other points were derived (Gordon *et al.* 1992). The theodolite used was electronic, thus the coordinates of all the points along a cross-section were calculated automatically. The cross-sections were surveyed in a straight line between two terminal points. The survey system (points) was fixed to the National Survey Coordinate system using a Global Positioning System (M.C. Briers, Surveyor - Geomatics Department, University of Cape Town, pers. comm. 2001).

The theodolite was also used to delineate the channel and create, plan and contour maps when the sites were surveyed the first time. Tape measures were used to delineate and create channel outlines when the theodolite was not available. The tape measures were laid in a straight line along the length of the channel. Channel widths were measured at right angles to this tape by using a second tape. Substratum and flow maps were also drawn at the same time while the tapes were still in place, using the method described by King & Schael (2001) (Section 13.4.2). The surveyors used the Computer Aided Drafting (CAD) programmes Autocad and Allycad to generate the plan maps and cross-sections. The drawn maps were scanned and digitised on screen with the aid of CorelDraw V 8.

13.3.2 Ecology: habitat maps

Habitat maps of substratum and flow types were drawn for Sites 1 to 3. There was little in the way of aquatic habitat diversity in the concrete channel in Site 4, which had mostly concrete with sand scattered on it. Flow types (Table 13.2) and substratum (Table 13.3) were visually identified in the field according to the categories given by King & Schael (2001). The different hydraulic biotopes (Table 13.4), which were combinations of substratum and flow type, were identified and sampled for aquatic macro invertebrates.

Table 13.2 Categories of visually distinct flow types identified in the study. (King & Schael 2001)

Flow type	Definition
Broken standing waves (BSW)	Standing waves present that break at the crest (white water).
Undular standing waves (USW)	Standing waves form at the surface with no broken water.
Rippled surface (RS)	Water surface has regular smooth disturbances forming low transverse ripples across the direction of flow.
Smooth boundary turbulent (SM)	Water surface remains smooth; medium to slow streaming takes place throughout the water profile; turbulence can be seen as the upward movement of fine suspended particles.
Barely perceptible flow (BF)	Smooth surface flow, only perceptible through the movement of floating objects.
No flow (NF)	No water movement

Table 13.3 Categories of substrata. (King & Schael 2001)

Category	Size range (mm)
Sand (SA)	0.063-2
Gravel (GR)	2-64
Cobble (C)	64-128
Boulder (B)	> 256

13.3.3 Ecology: macro-invertebrates

A modified version of the South African Scoring System Version 4 (SASS 4) biological monitoring method was used for the invertebrate sampling (Thirion *et al.* 1995). SASS 4 required specific time periods to identify the macro-invertebrate families. Longer time was taken in identifying the families because the researchers were not familiar with the different families. Sediment cores of approximately 0.2 m diameter were used to sample the sand bed under all flow types. A 250 μm mesh net was used to sample the marginal and in-channel vegetation by sweeping the net amongst and against the vegetation. Various sized samples were taken (Appendix 13.1). The net was also used to sweep the top parts of the sand where a core was not taken. The clayblock substratum, building rubble/wiring within the channel and cobbles were brushed with a commercial dishwashing brush and disturbed respectively to dislodge the organisms into the net.

Table 13.4 Hydraulic biotopes identified and sampled during summer 2001 and 2002. # = Reference number given to biotopes irrespective of flow types (See figures 13.10a, 13.11a and 13.12a). SA = Sand; G = Gravel; C = Cobble; B = Boulder; V = Emergent vegetation (growing in the channel and extending above the water surface); MV = Marginal vegetation (growing along the channel margins); SV = Submerged vegetation (submerged below the water surface); ODL = Organic debris and litter; BR = Building rubble. RS, SM, BF and NF are flow types (Table 13.2).

Hydraulic biotope descriptions	**# Hydraulic Biotope no.**	**Site 1**		**Site 2**		**Site 3**		**Site 4**	
		2001	**2002**	**2001**	**2002**	**2001**	**2002**	**2001**	**2002**
SA in SM	1			*	*	*			
SA+ODL in SM	2	*			*	*			
SA+ODL in BF	2			*			*		
SA+ODL+A in BF	3	*		*	*				
SA+G+ODL+A in SM	4		*		*		*		
SA+ODL+A+SV in SM	5				*				
SA+ODL+clay blocks in SM	6	*							
SA+SC+ODL+A+V in BF	7						*		
MV in SM	8			*		*			
MV in BF	8	*							
MV+A in BF	9		*	*	*	*	*		
V in SM	10								
V in RS	10	*							
V in BF	10			*					
V+A in NF	11			*		*			
V+A in BF	11					*			
SV in BF	12	*							
SV+A in SM	13				*				
C in SM	14								
C+A in SM	15			*	*	*	*		
C+G+ODL in SM	16					*			
C+B+ODL+A in SM	17						*		
B+ODL in BF	18					*			
Slope clay block+G+A in RS	19						*		
Clay block+G in RS	20	*							
BR +gabion wiring in SM	21			*					
Concrete+SA+G in SM	22						*		
Concrete in SM	23							*	*
Concrete+A in SM	24							*	*
Concrete+A in BF	24							*	*
Concrete+SA in SM	25								*
Concrete+SA+A in SM	26							*	*
Concrete+V+ODL+A in SM	27							*	*
Hole in concrete in SM	28							*	*

The samples were taken within each hydraulic biotope with the positions marked on the flow maps. The organisms were placed in a sorting tray and identified in the field using the SASS 4 user manual (Thirion *et al.* 1995). Families were assigned an abundance rating of 1 to 5 (King & Schael 2001) (Table 13.5). The samples were also fixed and preserved with 4% formalin and 80% ethanol respectively. Preservation of the organisms allowed for further identification in the laboratory and the building of a reference collection.

Table 13.5 Abundance ratings for macro-invertebrates. (King & Schael 2001).

Abundance Ratings	Numbers of animals per sample
1	1
2	2-6
3	7-20
4	21-100
5	>100

The invertebrate data were entered onto an Excel spreadsheet for analysis in Primer V5 (Clarke & Warwick 1994; Clarke & Gorley 2001). The Bray-Curtis similarity coefficient was used to generate a matrix of the similarity between samples. Prior standardisation of the data was necessary since not all the samples were of the same size. This was done by selecting the Standardise option. The data were also transformed to Presence/Absence format to give equal weight to all the families, whether rare or abundant. Cluster analysis and Ordination through Multi-Dimensional Scaling (MDS) were then performed on the data. The cluster analysis produced a dendrogram in which groups of similar samples cluster together. The ordination analysis produced a map of the samples in 2-dimensional space (MDS plot), with similar samples close together and dissimilar ones far apart. The stress required to produce the plot indicates how accurately the high-dimensional relationships among the samples are represented in 2-dimensional space (Clarke & Warwick 1994).

13.3.4 Ecology: vegetation

The vegetation was sampled along selected cross-sections at each site. First, a measuring tape was laid along the cross-section and then a $1m^2$ grid was used to delineate quadrats for sampling (Table 13.6). The vegetation was sampled in a non-destructive manner based on floristic characteristics. In each quadrat, the cover of each species was defined as the proportion of the ground occupied by a perpendicular projection of the aerial parts of each species, expressed as a percentage. The height of the tallest individual of each species was measured, together with a measure of the vigour of the species from 0 % (dead) to 100 % (very healthy). Vegetation samples were collected from the surrounding areas for identification (Goldsmith *et al.* 1986).

Vegetation data were entered into Turboveg V1.99b, a vegetation database (Botany Department, Stellenbosch University 2001) and then exported into an Excel spreadsheet. Primer V5 was then used to search for species groupings, as described for the macro-invertebrates, but the data were not transformed or standardised. The reason for this was that all the samples (quadrats) were the same size. Quadrats without vegetation (e.g. 100% bare ground) and outliers were excluded from the PRIMER analysis. Outliers were outlying points or samples that lay far apart from the rest of the data set and had a disproportionate effect on the data. These were removed to allow better examination of the remaining clusters.

Table 13.6 The number of vegetation quadrats surveyed per cross-section.

Site	Cross section	Number of quadrats 2001	2002
1	1.1	30	30
	1.2	24	30
	1.3	23	26
2	2.1	43	33
	2.2	40	42
	2.3	45	45
3	2.4	45	47
	3.3	46	50
	3.5	43	43
4	4.1	38	39

13.3.5 Discharge

The winter sampling programme entailed field observations during rain events, starting on the first day of rain with depth readings at Site 4 cross section 4.1. The field observations continued until it was safe to take discharge readings in all the sites. Discharge readings were taken on three dates during the winter. The depth measurements and discharge readings were used by Mr Jateen Bhana, a fourth year civil engineering student at the University of Cape Town, for his fourth year project. The aim of his project was to produce a discharge-stage rating curve for Site 4, cross-section 4.1. Detailed plan, substratum and flow maps were drawn when it was safe to do so. A photographic record was also compiled.

The velocity-area method was used to calculate discharge. A pygmy-type current meter was used to measure velocity. A top-setting wading rod was used to facilitate setting the current meter at 60% depths (Gordon *et al.* 1992).

13.4 Results

Presented here are examples of the kinds of data collected, with the full data set and analyses in Ms Fisher's M.Sc. thesis.

The cross-sections and habitat maps of Site 1 are used as an example to describe the changes that took place in the channel (Sections 13.4.1 and 13.4.2) due to the high discharge during the high winter rainfall of 2001. This is followed by presentation of the macro-invertebrate data (Section 13.4.3) collected over the period 2001-2002 at Sites 1 to 3, and the vegetation data for transect 1.2 (Section 13.4.4). Results were mostly from Sites 1 and 2 since major changes occurred at these sites.

Site 2 was channelised after the initial data collection exercise, in January and February of 2001, by straightening the channel and widening the banks (Figure 13.4). Bank erosion occurred within the widened section of Site 2 and between Sites 1 and 2 during May 2001 with the beginning of the winter rains. The Western Cape had unusually high rainfall from May to September 2001 (Figure 13.3). Further extensive bank erosion occurred within Sites 1 and 2, upstream of Site 1 and on the outer bends of meanders between Site 1 and 2 during the July 2001 floods. The bank erosion resulted in channel widening and deposition of eroded sediment in the channel further downstream. Site 3, not indicated on Figure 13.4, was also heavily silted up due to upstream bank erosion. The owner of the horse farm adjacent to Site 1 lost about 250 m^2 of land during the floods. This is equivalent to about 1250 m^3 of bank sediment entering the river from the right bank alone. The Oostenberg Municipality re-structured the banks to replace his land, using sediment from the channel. After the July 2001

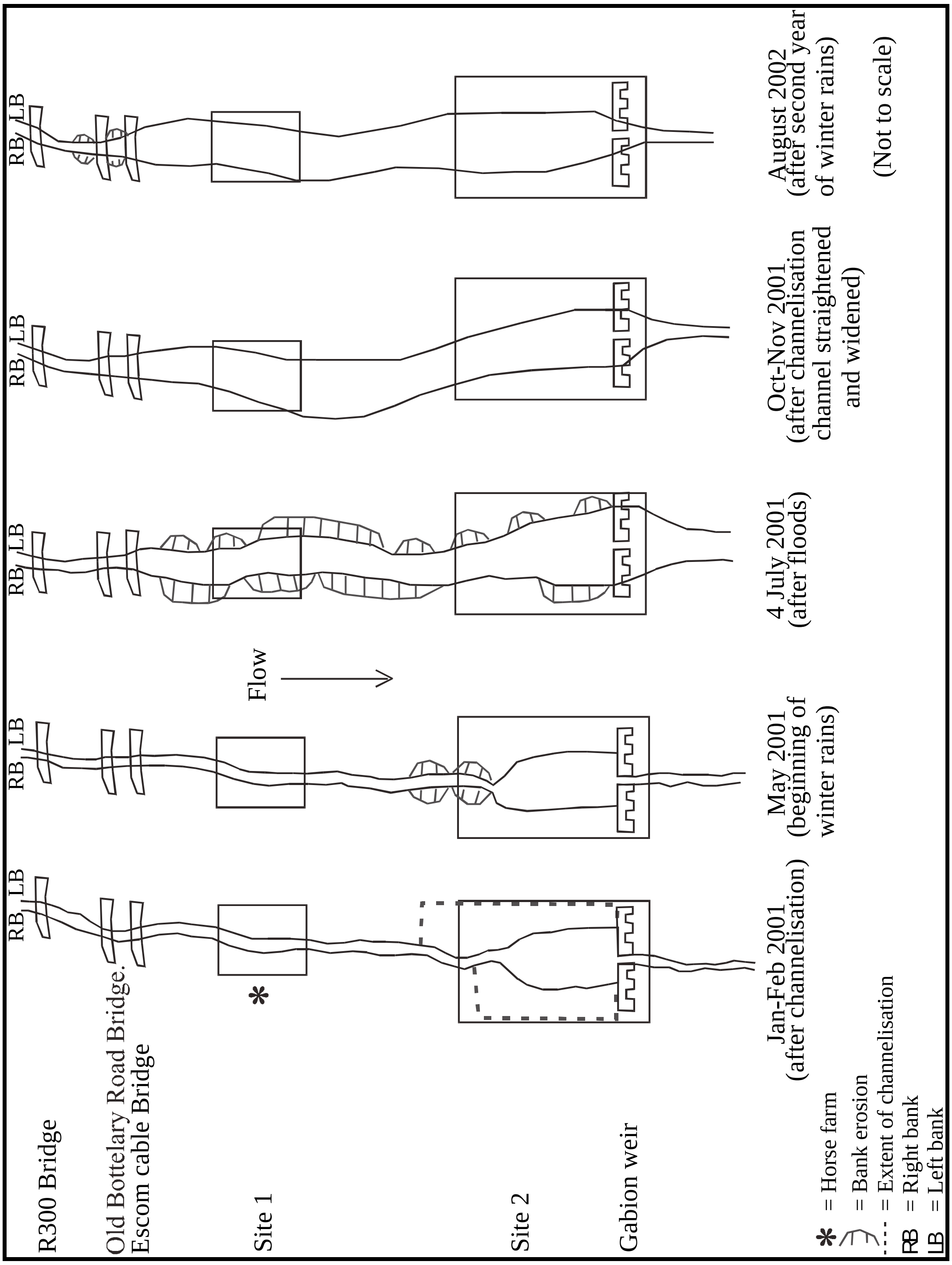


Figure 13.4 Channel changes through the study period as a result of the channelisation and winter floods.

floods, the channel was straightened and widened during October and November 2001 (Figure 13.4 and 13.5), from Site 2 upstream past Site 1 towards the Old Bottelary Road Bridge. This was done to eradicate the slight meandering of the channel in order to prevent future erosion on the outer bends and to accommodate floods of similar or larger magnitude. Those reaches straightened during

October and November 2001 and referred to above remained stable during the 2002 winter rainfall period, but bank erosion occurred at the Old Bottelary Road Bridge upstream of Site 1 with undercutting of bridge pillars. The impacts of the channelisation are discussed in Section 13.5.1.

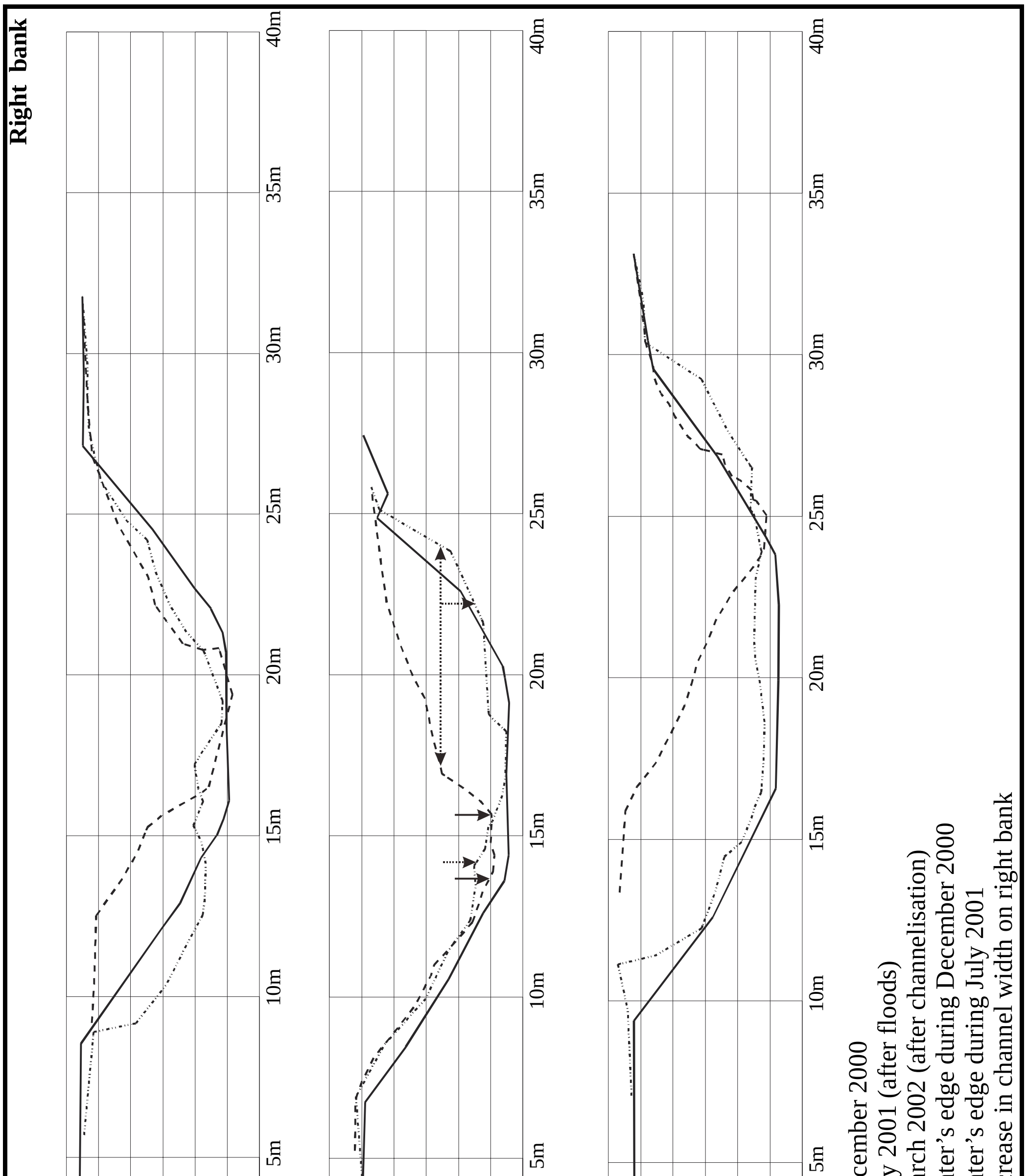


Figure 13.5 Cross-sections of Site 1 during the first and second summer after channelisation and intervening winter floods.

13.4.1 Site 1: cross-sections

The cross-sections of Site 1 (Figure 13.5) indicate that the channel was fairly narrow at the beginning of the study period (December 2000), with an active channel or thalweg slightly incised into a bigger macro channel. The definition of the active channel was more pronounced on the right bank (looking downstream) than on the left bank. The bank collapse during July 2001 resulted in more than 5 m of land being washed away on the left bank at cross-sections 1.1 and 1.3. The active channel of cross-section 1.2 widened by 2.5m and the midslope of the right bank (macro-channel) receded by 6.5m (indicated by arrows on Figure 13.5). The bank collapse also resulted in sediment being deposited in the channel as recorded at cross-section 1.1. The channelisation during October and November 2001 was an attempt to stabilize the banks of the river by re-grading them to an angle of about thirty degrees and furthermore resulted in a uniform straight channel. The channel was filled in on the right bank at cross-sections 1.2 and 1.3 and on the left bank at cross-section 1.1, resulting in a more or less trapezoidal cross-section.

13.4.2 Site 1: habitat maps

The following sections provide an overview of the substratum and flow types present in the study area during the study period and how they changed due to the floods and channelisation. The first section deals with the area at the start of the study period in February 2001. Section two covers the period during and shortly after the winter floods in July 2001 while the last section addresses the channel in January 2002 after channelisation was completed. The flow types are defined in Table 13.2 and hydraulic biotopes in Table 13.4.

Initial habitat – February 2001

At the beginning of the study period, Site 1 was approximately 4 m wide at cross-section 1.1, <2 m at cross-section 1.2 and 2 m at cross-section 1.3 (Figure 13.6a). The active channel meandered slightly within the macro-channel. Marginal vegetation and organic litter and debris occurred throughout the active channel. A pool with submerged vegetation occurred at the upstream end of the site near cross-section 1.1. From there, downstream for about 12 m, the channel bed consisted of sand, organic debris and litter with gravel patches. From about 10 m upstream of cross-section 1.2 to a downstream position halfway between cross-section 1.2 and 1.3, the middle of the channel was dominated by a semi-resistant organic clay layer referred to as “clayblock material” and gravel. The clayblock material in the channel often had a brown colour while that on the sides of the channel was usually grey or light coloured. A pool with sand, organic debris and litter as well as clayblock material and gravel occurred over the last 10 m at the downstream end of Site 1.

Typical summer base flow conditions characterised the flow patterns of Site 1 at the start of the study period. Barely perceptible flow dominated 79% of the site, not only in the upstream 15 m and downstream 7 m of the site, but also where marginal vegetation occurred. Rippled surface flow was associated with the clayblock and gravel substrata of the middle of the channel and accounted for about 7% of the flow types. Smooth flow occurred on the upstream and downstream sides of the rippled surface flow covering about 14% (Figure 13.6b).

Habitats - July 2001

The habitats mapped during the winter of 2001 (Figure 13.7 & 13.8) showed a completely different picture compared to those mapped during the previous summer. The winter floods resulted in bank erosion and consequent increased meandering of the active channel. The active channel at cross-section 1.1 widened from 4 to 7.6 m, from <2 to 7.6 m at cross-section 1.2 and from 2 to 7.6 m at cross-section 1.3. The widest section of the channel was halfway between cross-section 1.2 and 1.3 where the channel was 15.5 m wide. Marginal vegetation was found infrequently with patches of

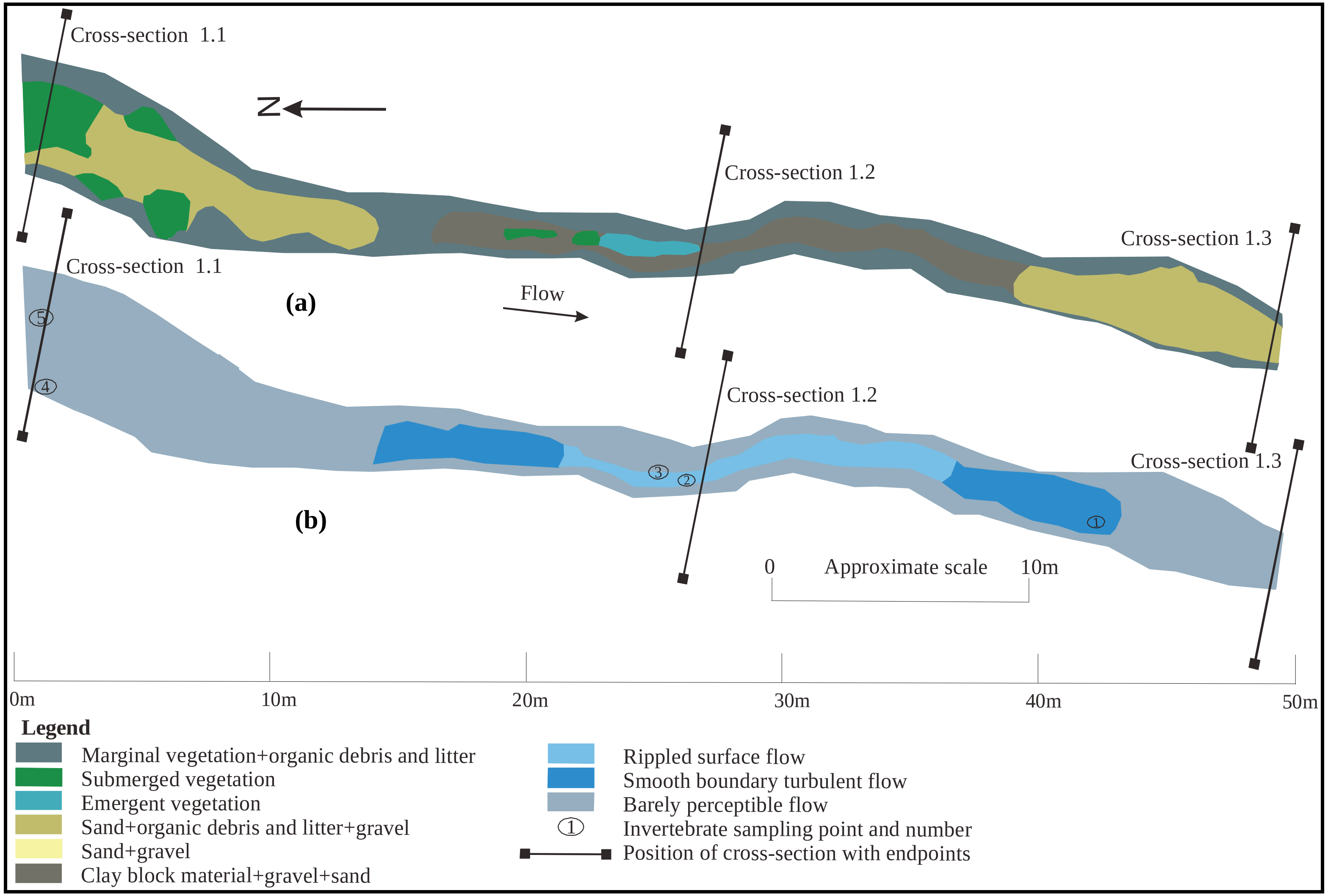


Figure 13.6: Substratum (a) and flow type (b) maps of Site 1 - February 2001.

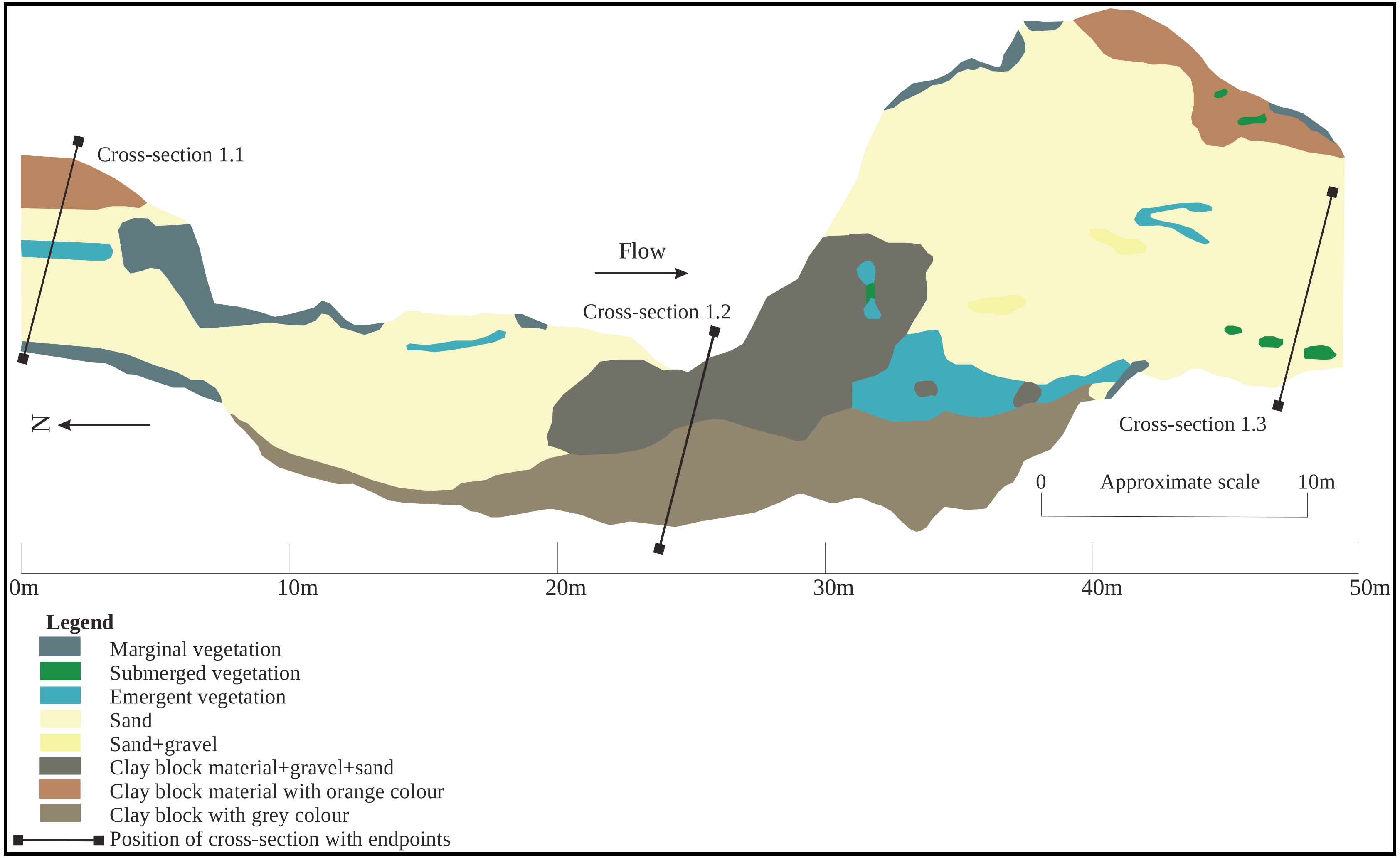


Figure 13.7: Substratum map of Site 1 after floods (July 2001).

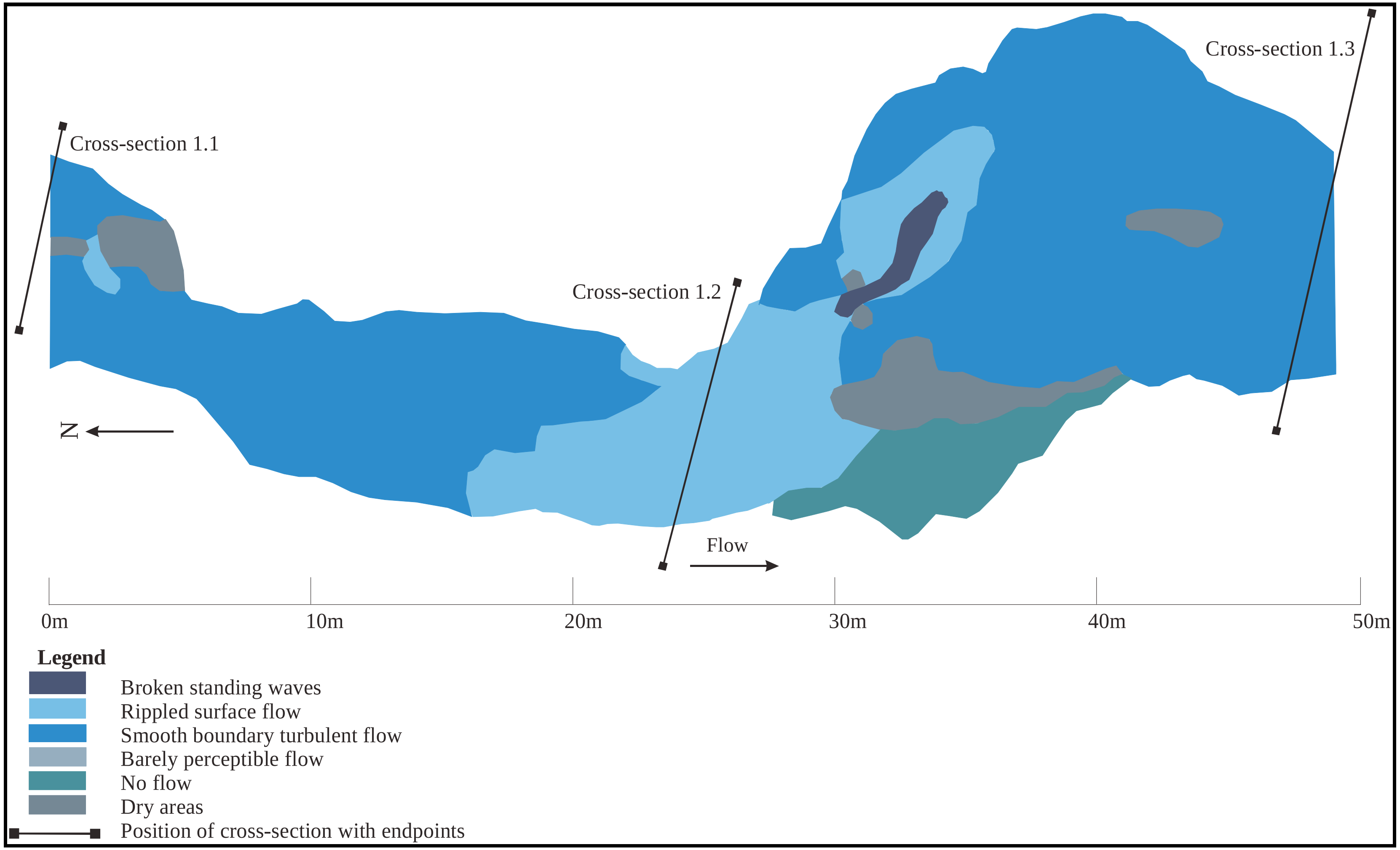


Figure 13.8: Flow map of Site 1 after the floods (July 2001).

emergent and submerged vegetation. The bank erosion on the right bank near cross-section 1.2 exposed the more resistant clayblock material, referred to earlier, at the bottom of the slope. The exposed clayblock material extended from about 6 m downstream of cross-section 1.1 to about halfway between cross-sections 1.2 and 1.3. The same type of resistant clayblock material also occurred on the left bank near cross-section 1.1 and 1.3. The channel itself was mostly sand-covered with occasional small gravel patches. Sand also covered the resistant clayblock material in places (Figure 13.7).

The winter flow pattern (Figure 13.8) was also more diverse than mapped previously in February 2001. The site consisted of 66 % smooth boundary turbulent flow. Some of the marginal and emergent vegetation nearest to the left bank was dry. These dry patches formed a restriction to the flow and resulted in rippled surface flow between the dry areas and beyond them. The dry areas made up 6 % of the channel area, while the exposed clayblock material had barely perceptible flow, covering about 6 %. The vegetation downstream of the clayblock material halfway between cross-sections 1.2 and 1.3 (Figure 13.7) was dry. The two vegetation clumps near the left bank and just downstream of cross-section 1.2, also formed a restriction forcing the flow over the submerged vegetation and between the two dry vegetation patches. This resulted in broken standing waves with rippled surface flow further downstream. Broken standing waves made up 2 %, while rippled surface flow made up 20 % of total channel area. The vegetation patch in the middle of the channel upstream of cross-section 1.3 was also dry.

Habitats - January 2002

The restructuring of the banks and straightening of the channel bed between October and November 2001 resulted in a fairly straight and wide (± 5.5-7 m) active channel (Figure 13.9a). Vegetation, organic debris and litter and algae were found along the channel margins. The channel bed consisted of sand+gravel+organic debris and litter+algae. A patch of iron stained sand+gravel+organic debris and litter+algae occurred in the middle of the channel about 5 m upstream of cross-section 1.3. Vegetated sand bars occurred infrequently along the channel margins. The reduction in channel width downstream of cross-section 1.2 was the result of a small alluvial fan that formed along the left bank due to the slope failure resulting from seepage that followed abnormally high rainfall in early January 2002.

There was very little flow diversity (Figure 13.9b) after the channelisation. Barely perceptible flow was associated with the marginal vegetation, while the vegetated sand bars were dry. Smooth boundary turbulent flow was found along the centre of the channel and there was also a small area with no flow upstream of cross-section 1.3. Barely perceptible flow made up 15 %, smooth boundary turbulent flow 80 %, dry areas 4.8 % and no flow 0.2 % of the total site area.

In summary, the channel was confined initially within a macro-channel. Semi-resistant clay block material dominated the centre of the channel with sand, organic debris and litter in patches and marginal vegetation along the margins. Flow types consisted of barely perceptible, smooth boundary turbulent and rippled surface flow with barely perceptible flow dominating. Bank collapse due to the floods resulted in the exposure of resistant bank material, widening and the silting up of parts of the channel. These together with increased discharge associated with the floods, led to more diverse flow types. Finally channelisation resulted in a uniform, straight channel with a decrease in substratum and flow types.

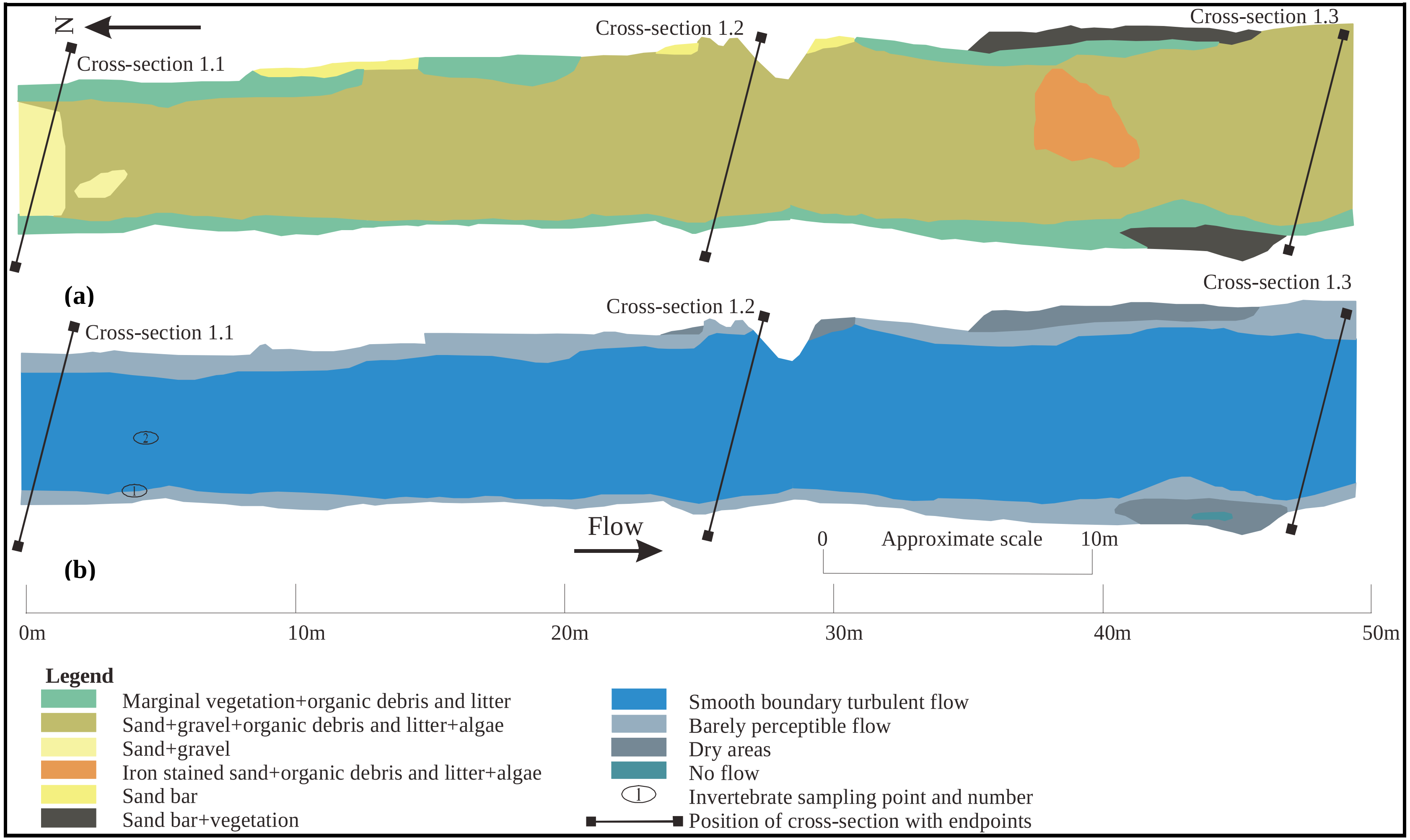


Figure 13.9: Substratum (a) and flow type (b) maps of Site 1 after channelisation (January 2002).

13.4.3 Macro-invertebrates

The cross-sections and habitat changes documented in sections 13.4.1 and 13.4.2 respectively impacted on the composition of macro-invertebrate assemblages in the study area. Data on these assemblages in both sampling periods were analysed together per site (Figure 13.10-13.12). A full list of the sample codes used, sample size, community compositions, and abundance ratings is given in Appendix 13.1.

Site 1 – 2001&2002

The dendrogram (Figure 13.10a) separated the samples into two groups with 20% similarity. The first group of samples was collected in emergent vegetation. Sample S1-1-02 was not closely related to the other two samples in the group (Figure 13.10b) because it consisted of Coenogrionidae, Dytiscidae, Veliidae, Naididae and Physidae and only had Physidae and Coenagrionidae in common with the other samples. The second group consisted of samples were collected in a mixture of biotopes including sand, emergent vegetation and clay block material (Table 13.4). The first two of these formed a pair at 80% similarity and had Chironomidae and Naididae in common. Sample S1-2-02 contained Physidae in addition, while sample S1-1-01 contained Naididae only.

The number of biotopes sampled decreased drastically during 2002 (Table 13.4) because the channelisation resulted in a very uniform channel-bed with only two biotopes available. The family composition remained similar between the years (Tables 13.7 and 13.8), but some families present during 2001 were absent during 2002 and vice versa. There were fewer families in 2002 compared to 2001 (Table 13.8).

Site 2 – 2001&2002

Three groups were identified on the dendrogram and MDS plot (Figure 13.11). The first group consisted of samples collected from sand and contained Naididae only. The two samples were 100% similar. The second group consisted of samples collected in a biotope where sand was the main substratum (Table 13.4). Invertebrates found included Physidae, Chironomidae, Simuliidae and Naididae. All the samples in this group had Physidae in common. Group 3 consisted of a mixture of biotopes sampled including marginal vegetation, emergent vegetation, building rubble, gravel+sand, sand+organic debris and litter+algae and sand+organic debris and litter+algae+submerged vegetation. Sample S2-2-01 was not closely related (Figure 13.11b) to the other samples in the group because it only contained five of the thirteen families found in this group (Appendix 13.1). All the samples in this group had Coenogrionidae in common.

Groups 2 and 3 were more closely related to each other than to Group 1 because the latter consisted of samples that contained Naididae only. Tables 13.7 and 13.8 show there was a decrease in community composition and abundance ratings during 2002, except at Site 4. The number of biotopes sampled also decreased from nine in 2001 to eight during 2002 except for Site 4 where it remained the same (Table 13.4).

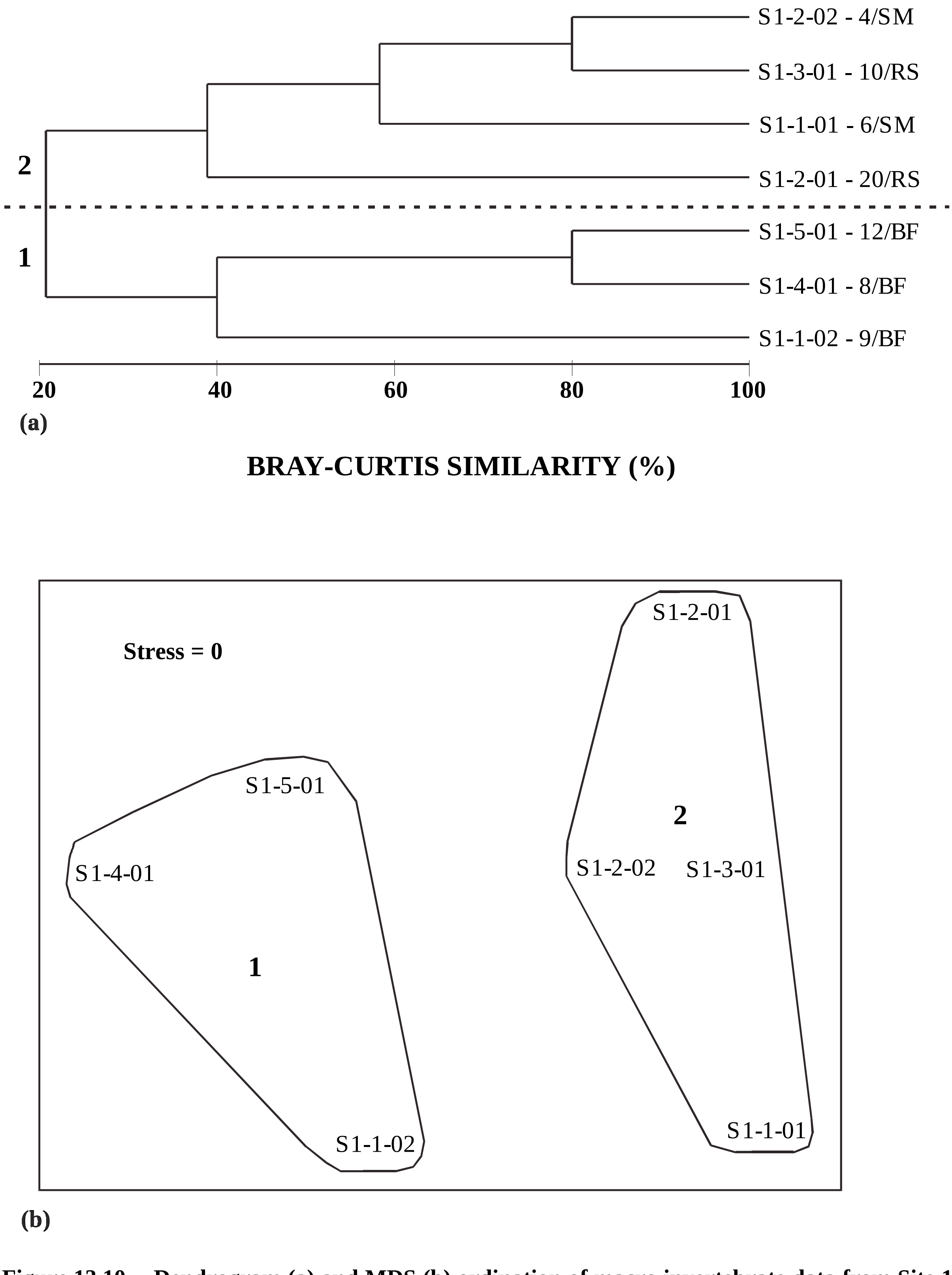


Figure 13.10 Dendrogram (a) and MDS (b) ordination of macro invertebrate data from Site 1 for 2001 and 2002. Code: S1-1-01 – 6/SM = Site, sample number, year – hydraulic biotope number (Table 13.4) and flow type (Table 13.2).

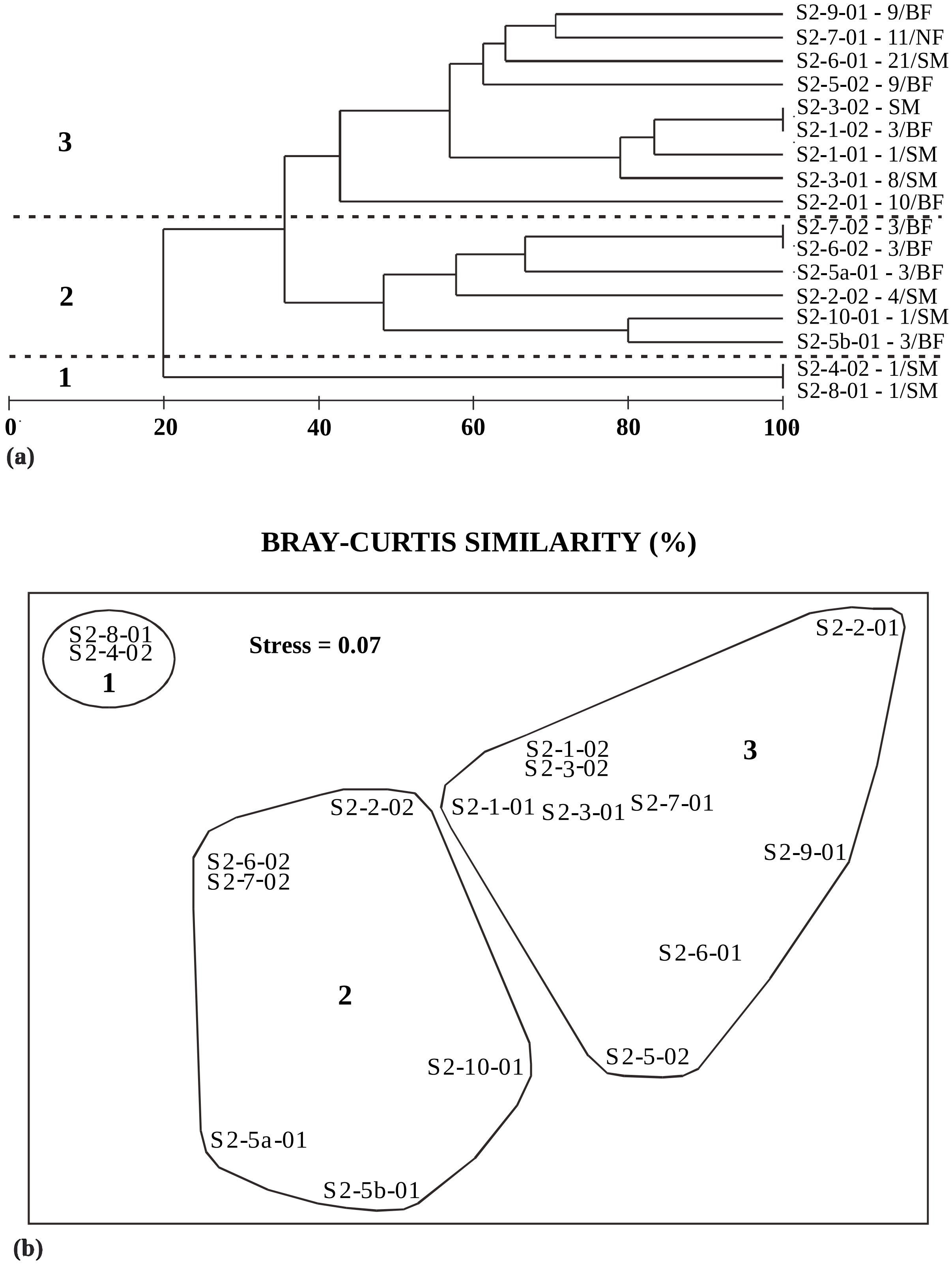


Figure 13.11 Dendrogram (a) and MDS (b) ordination of macro invertebrate data from Site 2 for 2001 and 2002. Code: S2-1-01 – 6/SM = Site, sample number, year – hydraulic biotope number (Table 13.4) and flow type (Table 13.2).

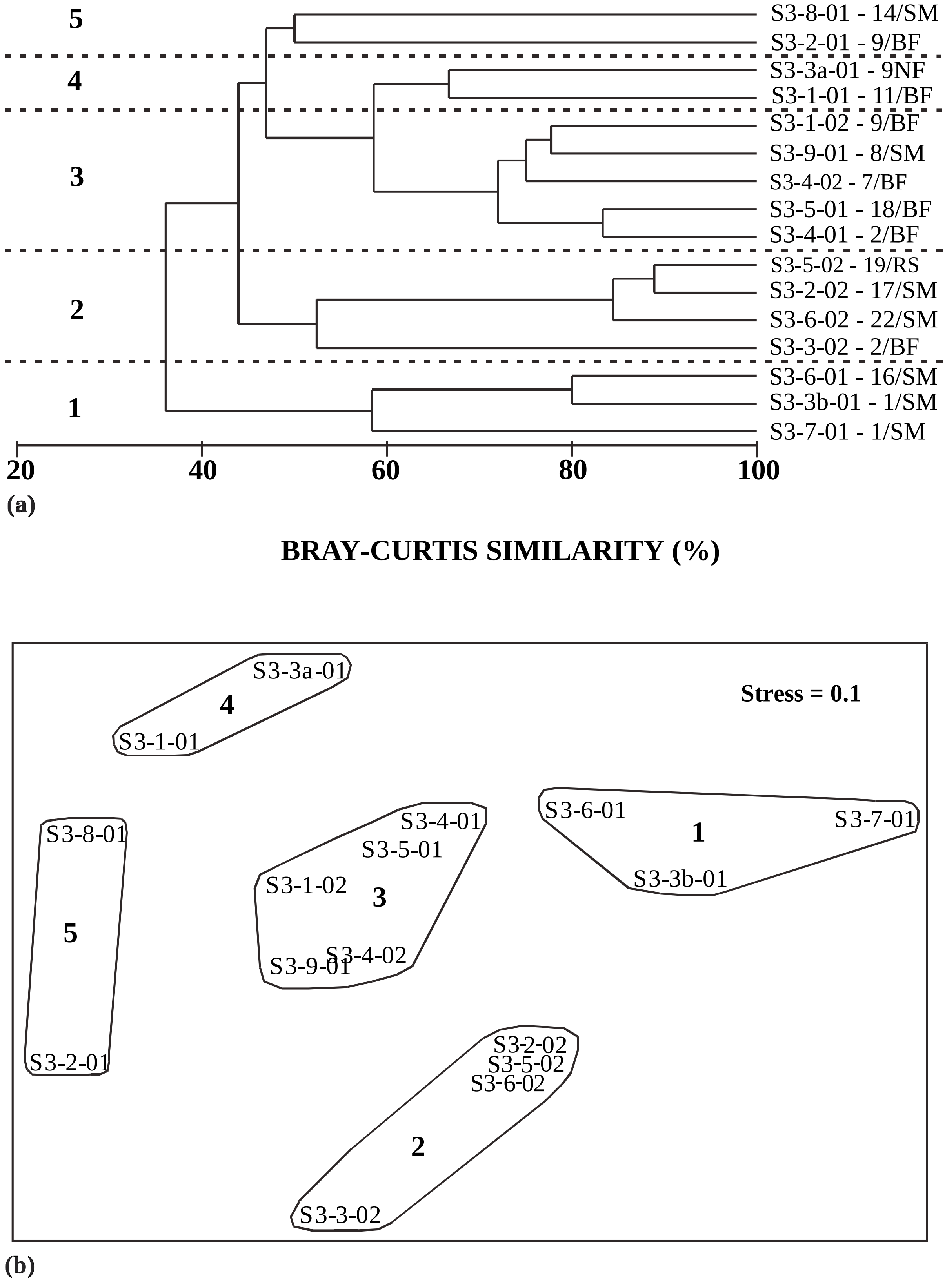


Figure 13.12 Dendrogram (a) and MDS (b) ordination of macro invertebrate data from Site 3 for 2001 and 2002. Code: S3-1-01 – 6/SM = Site, sample number, year – hydraulic biotope number (Table 13.4) and flow type (Table 13.2).

Site 3 – 2001&2002

Five groups were formed (Figure 13.12). Group 1 was formed at 57% similarity. The samples were taken in sand+organic debris and litter, and cobbles+organic debris and litter+algae. The families found included Naididae, Physidae and Corixidae. Group 2 consisted of samples collected in cobble+boulder+organic debris and litter+algae, gravel+slope clayblock material+organic debris and litter, a concrete slab, and sand+organic debris and litter. Sample S3-3-02 was 57% similar to the rest of the samples in the group. The remaining samples were 85% similar to each other. Samples S3-2-02, S3-5-02 and S3-6-02 had Chironomidae, Naididae, Physidae and Simuliidae in common with sample S3-2-02 having Dytiscidae in addition and sample S3-6-02 having Coenogrionidae in addition. Sample S3-3-02 contained Gomphidae, Chironimidae and Physidae. Group 3 consisted of samples taken in a mixture of biotopes including sand+organic debris and litter, marginal vegetation, marginal vegetation+algae, and boulder+organic debris and litter. The samples had Physidae and Corixidae in common (Appendix 13.1). The samples in Group 4 were taken in marginal and emergent vegetation and algae biotopes. These samples contained Corixidae and Physidae, but had Baetidae in addition. Samples S3-2-01 and S3-8-01 made up Group 5, which were sampled in marginal vegetation and cobbles respectively. The samples had Gerridae, Baetidae, Physidae and Hirudinae in common, but were not closely related.

In summary, there seemed to be no distinction between the sites with similar invertebrate families found in all the sites. There also seemed to be no distinction according to biotopes sampled as certain groups consisted of a variety of biotopes sampled. Site 1 was an exception, as there was a separation of samples from vegetation biotopes from those collected in sandy biotopes. The channelisation resulted in a loss of available habitats during 2002 and thus a decrease in community composition and abundance in most sites, except Site 4, which was already low due to it being a concrete canal.

Table 13.7 Summary table of invertebrate families present in all sites during 2001 and 2002.

Invertebrate	**Site 1**		**Site 2**		**Site 3**		**Site 4**	
Families	**2001**	**2002**	**2001**	**2002**	**2001**	**2002**	**2001**	**2002**
Hirudinae			*	*	*			*
Naididae	*	*	*	*	*	*	*	*
Coenogrionidae	*	*	*	*	*	*		
Gomphidae	*		*	*	*	*	*	
Aeshnidae					*			
Libellulidae			*					
Baetidae	*		*	*	*	*	*	*
Corixidae			*		*	*		
Pleidae								*
Naucoridae								*
Veliidae		*		*	*			
Gerridae			*		*	*		
Gyrinidae			*					
Dytiscidae		*	*	*		*		*
Simuliidae			*	*	*	*		*
Chironomidae	*	*	*	*	*	*	*	*
Culicidae			*		*			
Athericidae								*
Lymnaeidae	*		*	*	*	*		
Physidae	*	*	*	*	*	*	*	*
Total	**7**	**6**	**15**	**11**	**14**	**11**	**5**	**10**

Table 13.8 Summary table of invertebrate abundance ratings for all sites during 2001 and 2002.

Invertebrate	Site 1		Site 2		Site 3		Site 4	
Families	2001	2002	2001	2002	2001	2002	2001	2002
Hirudinae	0	0	2	10	8	0	0	1
Naididae	3	5	8	22	12	10	3	14
Coenogrionidae	5	2	11	6	4	3	0	2
Gomphidae	6	0	8	3	8	3	7	0
Aeshnidae	0	0	0	0	1	0	0	0
Libellulidae	0	0	2	0	0	0	0	0
Baetidae	4	0	3	3	13	7	3	5
Corixidae	0	0	10	0	16	3	0	0
Pleidae	0	0	0	0	0	0	0	2
Naucoridae	0	0	0	0	0	0	0	1
Veliidae	0	1	0	2	4	0	0	0
Gerridae	0	0	3	0	4	1	0	0
Gyrinidae	0	0	3	0	0	0	0	0
Dytiscidae	0	3	1	3	0	8	0	0
Simuliidae	0	0	3	2	8	6	20	4
Chironomidae	12	2	26	15	12	21	21	21
Culicidae	0	0	1	0	1	0	0	0
Athericidae	0	0	0	0	0	0	0	2
Lymnaeidae	1	0	8	2	10	3	0	2
Physidae	5	6	22	17	22	18	4	14

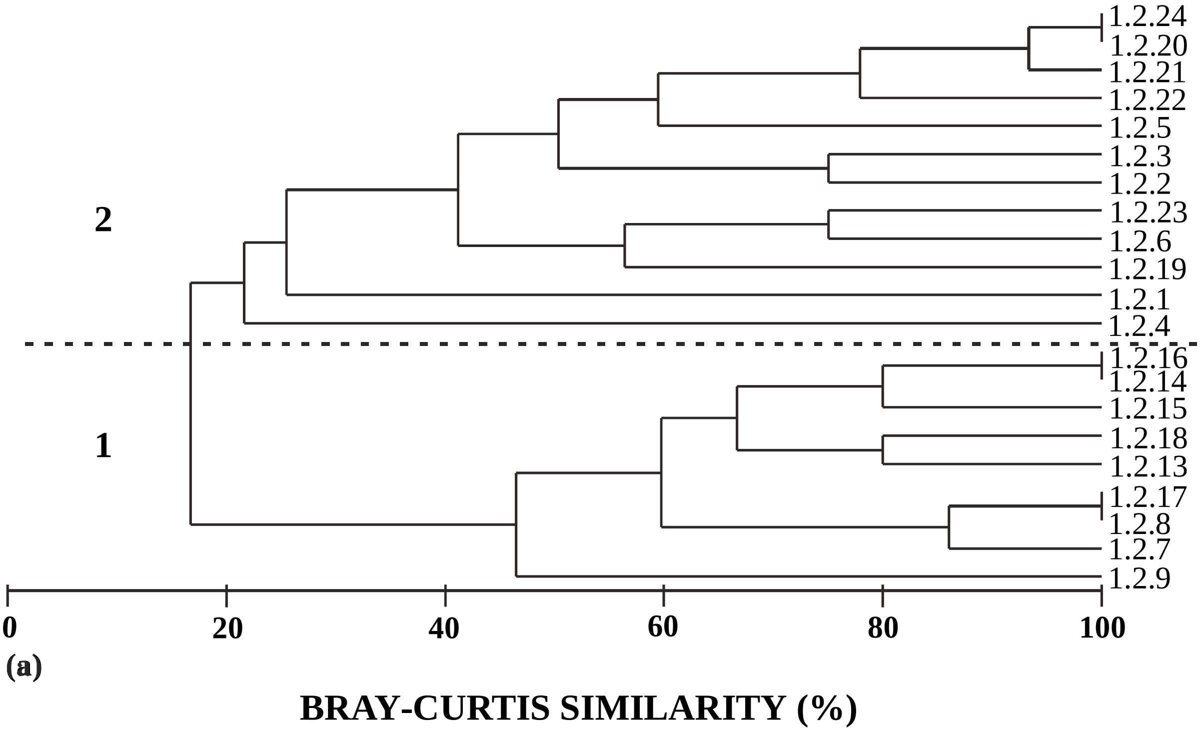
13.4.4 Vegetation

The changes documented in the cross-sections (13.5.1) had a profound effect on the plant assemblages. During 2001, the banks were covered with *Pennisetum clandestinum* (kikuyu), annual weeds and a few alien trees. After the channelisation the banks were left bare with only a thin strip of *P. clandestinum* along the water's edge. A new set of plant species was found during 2002. This section outlines the plant assemblages found before (2001) and after the channelisation (2002).

Vegetation transect 1.2 - 2001

There were two distinct groups of quadrats on the dendrogram and MDS plot at 17% similarity (Figure 13.13a and b). Group 1 contained plant species *Bromus diandrus*, *Pennisetum clandestinum*, *Polygonum aviculare*, *Chenopodium album*, *Isolepis fluitans* and *Cyperus longus* (Appendix 13.2). The quadrats occurred along the lower parts of both banks and within the channel (Figure 13.14). Most of the species in this group were usually found in wetter conditions with the exception of *Bromus diandrus*, which was tolerant to drier conditions (Hawaiian Ecosystem at Risk Project 2003, www.hear.org). This group of species was termed the wet-bank vegetation (Boucher 2002).

The second group consisted of *Bromus diandrus* and *P. clandestinum* of which the former was adapted to drier conditions as was indicated earlier while *P. clandestinum* was common throughout the study area. The quadrats occurred from the top of the bank to approximately 6 m down the slope on both banks (Figure 13.14).



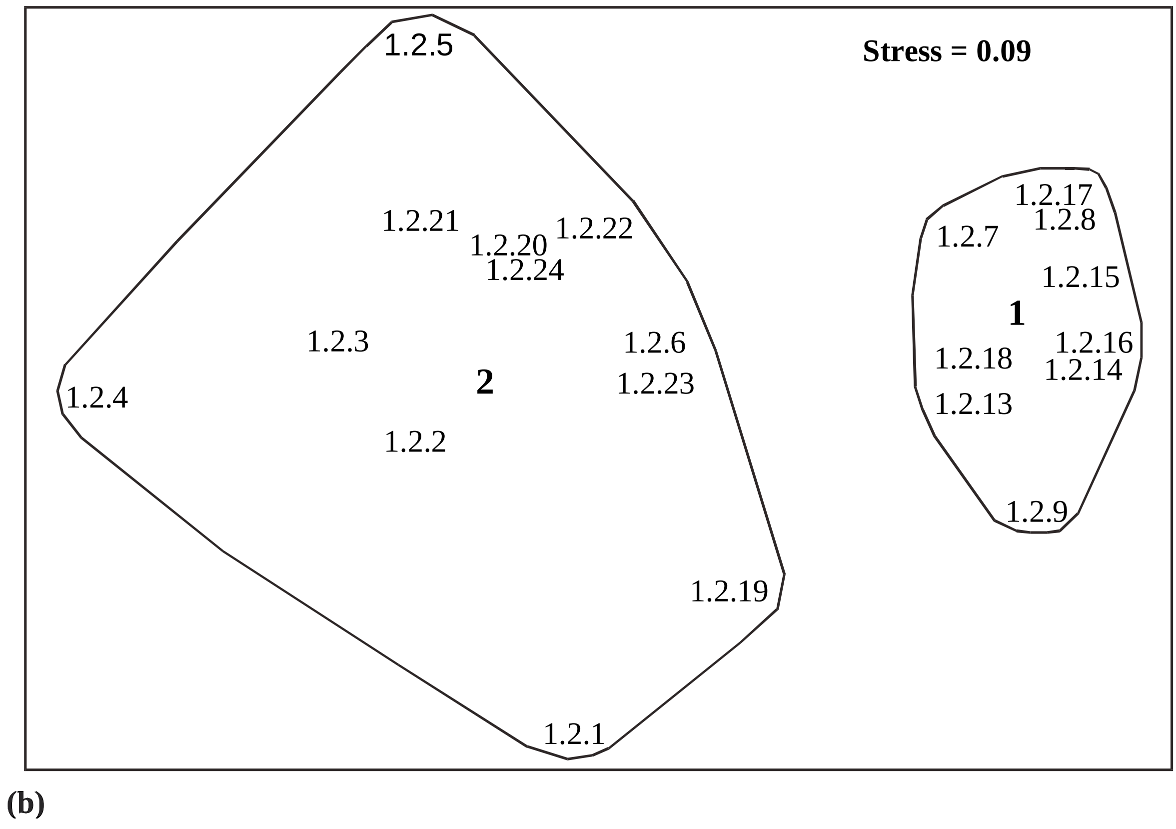


Figure 13.13 **Dendrogram (a) and MDS (b) ordination of vegetation data from vegetation transect 1.2 for 2001 showing the two groups identified.**

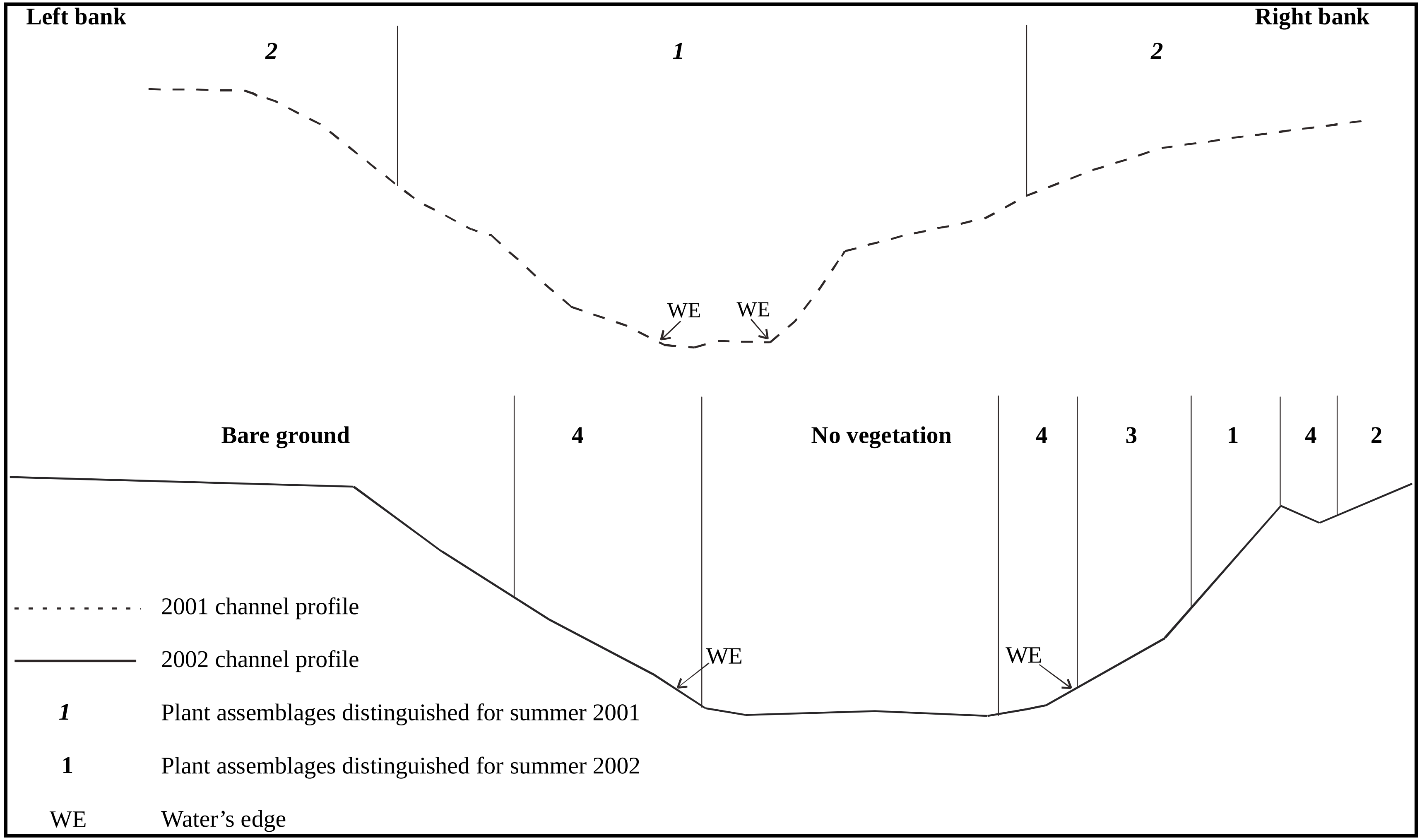


Figure 13.14 Plant assemblages and their position on vegetation transect 1.2 for 2001 and 2002.

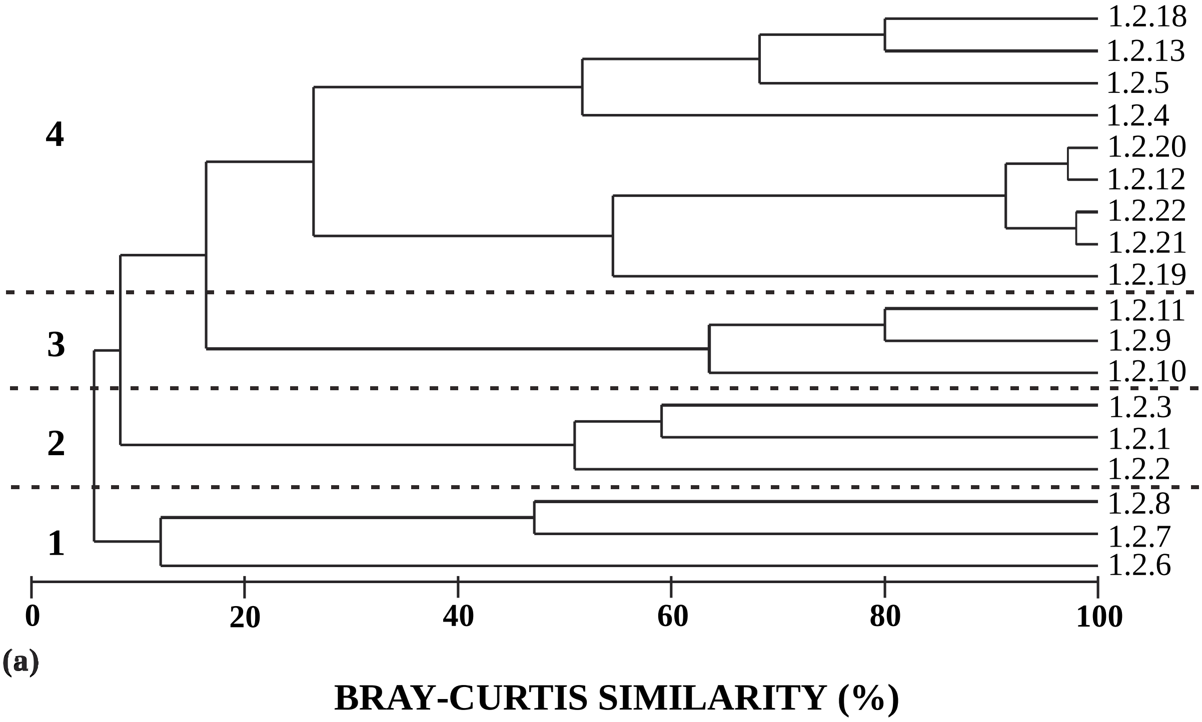
Vegetation transect 1.2 – 2002

There were four plant groups identified on the dendrogram and MDS plot (Figure 13.15a and b), but they were different to the groups identified in 2001 and with different plants. Group 1 consisted of a mixture of plant species including *Acacia saligna, Pennisetum clandestinum, Cynodon dactolyn, Brachiaria serrata*, *Cyperus rotundus*, *Datura stramonium*, *Citrullus lanatus* and *Tribulus terrestris*. This group occurred along the middle of the right bank slope (Figure 13.14). Group 2 occurred on top of the right bank (Figure 13.14) and contained *Echium plantagineum*, *Pennisetum clandestinum*, *Cynodon dactolyn*, *Amaranthus deflexus*, *Raphanus raphanistrum* and *Prunus* species. Group 3 occurred along the lower parts of the right bank and close to the water's edge. Plant species included *Pennisetum clandestinum, Cynodon dactylon*, *Cyperus rotundus* and *Citrullus lanatus*. Group 4 consisted of *Pennisetum clandestinum* mainly, with a small percentage (2%) of *Cyperus rotundus* (Appendix 13.2) and occurred along the water's edge on both banks.

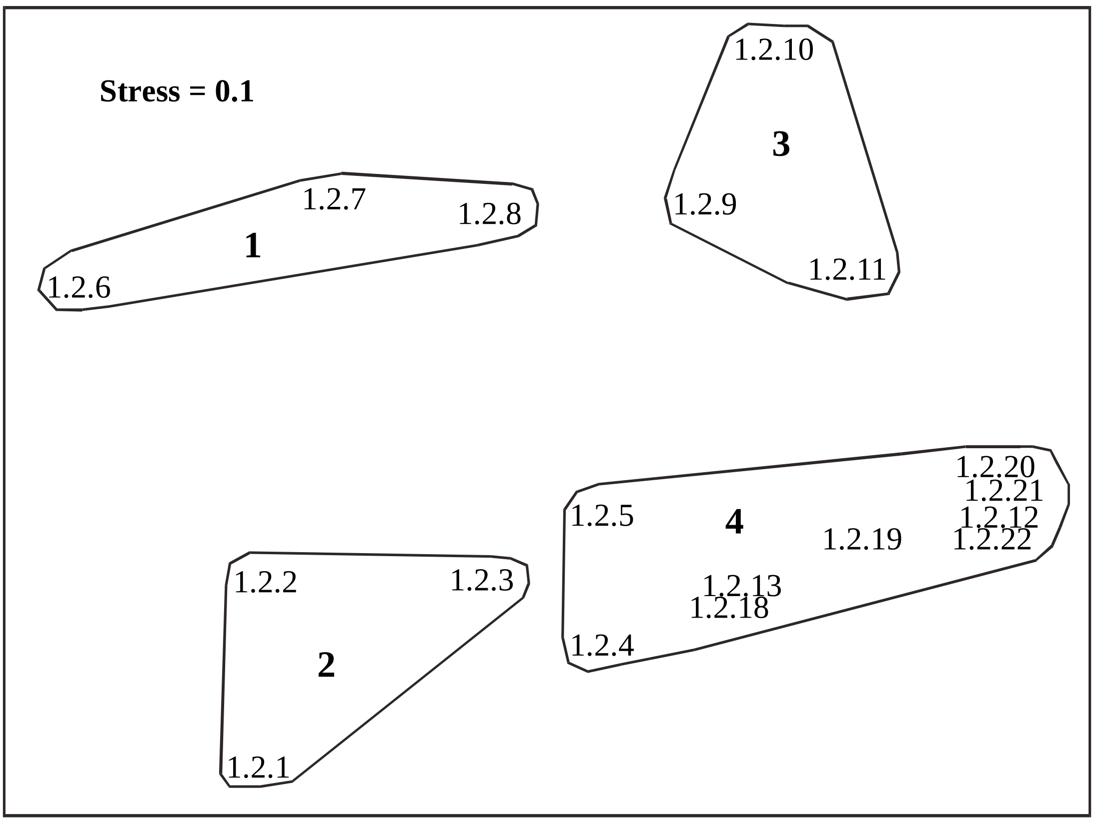
From the vegetation composition and distributions outlined above, it seems that plant species for 2001 fall in two groups, one of which can be classified as wet-bank vegetation, the other as dry-bank vegetation. They also occurred in distinct positions along the cross-section. More species and more groups occurred in 2002, but the pattern is less clear. Most of the species were alien weeds. Of the total number of species, 75% were weeds in 2001 and 87% in 2002 (Table 13.9).

13.4.5 Discharge

Discharge readings were taken to develop an understanding of how the river reacted to rainfall events. Discharge readings were taken on three occasions during the winter of 2001. On two occasions, May and July, this was done shortly after rainfall events. The May readings were taken after about 57 mm fell in the Durbanville area during the week prior to 11 May, while 183mm fell in the week prior to 12 July. Only 39 mm fell during the 27 days of June prior to the 28th of June (Unofficial record, A.C.T. Scheepers pers. comm. 2003) (Table 13.10). Discharge was calculated by averaging the two or three different readings taken through the study area on any one day. Discharge at Site 4 on 11 May was 0.091 $m^3 s^{-1}$ (Table 13.11). This increased to an average of 0.257 $m^3 s^{-1}$ by 28 June 2001. The highest reading was taken on 12 July 2001, when 0.756 $m^3 s^{-1}$ was measured at Site 3 (Table 13.11). No readings were taken on 3 July 2001 when the banks started to collapse at Sites 1 and 2, because flows were too high to access the river following heavy falls of 122 mm in the preceding three days in the Durbanville area (Unofficial record, A.C.T. Scheepers pers. comm. 2003) (Table 13.10). From the above-mentioned discharge events, it can be seen that the considerable volumes of water in the Kuils River after rains can erode the channel and banks, changing the course of the river.



(a)



(b)

Figure 13.15 **Dendrogram (a) and MDS (b) ordination of vegetation data from vegetation transect 1.2 for 2002 showing the four groups identified.**

Table 13.9 **Plant species present in all vegetation transects of Site 1 for 2001 and 2002, with growth patterns, origin, and weed status.** The table shows that the vegetation consisted mostly of alien weed species in both years, but there were more plant species in 2002 than in 2001. Where A = Annual, P = Perennial, B = Bi-annual, AL = Alien and IN = Indigenous (Bond & Goldblatt 1984; Bromilow 1995; Henderson 2001; Levyns 1966).

Species	Growth pattern	Origin	Weed
2001			
Bromus diandrus	A	AL	Yes
Echium plantagineum	B	AL	Yes
Pennisetum clandestinum	P	AL	Yes
Cleome monophylla	A	IN	Yes
Lolium temulentum	A	AL	Yes
Cynodon dactylon	P	IN	No
Polygonum aviculare	A	AL	Yes
Chenopodium album	P	AL	Yes
Cyperus species	P	AL	Yes
Isolepis fluitans	P	IN	No
Xanthium strumarium	A	AL	Yes
Cyperus longus	P	AL	Yes
2002			
Acacia saligna	P	AL	Yes
Echium plantagineum	A	AL	Yes
Pennisetum clandestinum	P	AL	Yes
Cynodon dactylon	P	AL	Yes
Cleome monophylla	A	IN	Yes
Isolepis fluitans	P	IN	No
Rumex species	A, P or B	AL	Yes
Cyperus denudatus	P	AL	Yes
Setaria species	A and P	IN	Yes
Sonchus oleraceus	A	AL	Yes
Amaranthus deflexus	P	AL	Yes
Brachiaria serrata	P	IN	Yes
Cyperus species	P	AL	Yes
Lolium perenne	P	AL	Yes
Polygonum species	A	AL	Yes
Raphanus raphanistrum	A or B	AL	Yes
Medicago polymorpha	A or P	AL	Yes
Tribulus terrestris	A	IN	Yes
Cyperus rotundus	P	AL	Yes
Datura stramonium	A	AL	Yes
Citrullus lanatus	A	IN	Yes
Euphorbia peplus	A	AL	Yes
Xanthium strumarium	A	AL	Yes
Carpha glomerata	P	AL	Yes

Table 13.10 Rainfall in mm measured in the Durbanville area prior to the discharge readings taken. * indicates the day when the banks started to collapse in Sites 1 and 2. No discharge readings were taken (Unofficial record, A.C.T. Scheepers).

Date of discharge reading	**Rainfall (mm)**
11 May 2001	57
28 June 2001	39
03 July 2001*	122
12 July 2001	183

Table 13.11 Discharge ($m^3 s^{-1}$) measured on three dates during winter 2001.

Location of measurement	**11 May 01**	**28 Jun 01**	**12 Jul 01**
Site 1			
Cross-section 1.1	0.083	0.396	0.749
Cross-section 1.3	0.098	0.505	0.729
Site 2			
2.5m on tape	0.090	0.396	0.529
61m on tape	0.089	-	0.588
Site 3			
28.4m u/s of pipe	0.106	0.383	0.756
10.2m u/s of gabion weir	0.099	-	-
14m d/s of gabion weir	0.067	0.547	0.732
Site 4			
Cross-section 4.1	0.087	0.090	0.622
Cross-section 4.2	0.098	-	0.670
Min	0.067	0.090	0.529
Max	0.106	0.547	0.756
Avg.	0.091	0.257	0.597

13.5 Discussion

13.5.1 Cross-sections and habitat maps

Factors that could be responsible for changes in the cross-sections include soil properties, vegetation cover, excessive and prolonged rainfall and downstream channelisation. The Kuils River has banks of cohesionless sand material with little clay. Bank collapse can occur through dislodgement of individual grains from the bank surface (bank face and base) by hydraulic action. The removal of base material leads to oversteepening and induces gravitational failure. Weakening and failure of the bank are exacerbated during heavy or prolonged rainfall events or rapid drawdown following high flows. Bank stability can fail due to a decrease in strength of the bank material and increase in unit weight, accompanying an increase in the degree of saturation. Floods provide higher than normal discharges with high flow velocities. The high-velocity flows exert greater shear stresses and macro turbulent flow at the bank surface. This can lead to failure, with an increase in channel width and depth (Kochel 1988; Wohl 2000). Field observations on 3 July 2001 revealed that lumps of bank material were being eroded at Site 2.

The most obvious channel adjustment due to floods is widening, which is often associated with a change in channel pattern, usually from single thread meander to braided channel (Eaton & Lapointe 2001). Where sediment supply is higher than the flow transport capacity, deposition can occur along the entire channel or in isolated reaches (Wohl 2000). The floods in the Kuils River caused extensive channel widening at Sites 1 and 2, with sediment deposited along the entire length of the channel downstream of the Old Bottelary Road Bridge. Similarly, the Eel River, California, USA experienced a 1-in-100 year flood during December 1964 (Patrick *et al.* 1982). The flood caused bank erosion and landslides, producing high sediment loads. Channel morphology was altered substantially with the sediment produced by the flood deposited in the channel, causing the riverbed to rise by 1.8-2.4 m.

Vegetation root systems reinforce bank material, imparting a greater resistance to erosion particularly in the upper bank. Grassy vegetation types have higher root densities, but woody plants have deeper and larger roots than grasses, and provide better protection against undercutting along large rivers (Friedman & Auble 2000). The banks of Site 1 and 2 were covered by non-woody plant species (weeds and annual grasses) with shallow root systems. The shallow root systems did not provide the necessary stability when the banks were saturated after the excessively high and prolonged rainfall.

Chapter Two of this report provides a comprehensive literature review on the impacts of channelisation on channel morphology. In summary, when a river is straightened the channel path is shortened. This increases the channel gradient, which in turn increases flow velocities in the straightened reach. The erosive capacity of the water is also increased, and can accommodate larger amounts of sediment supplied from upstream reaches. The imbalance in the sediment regime leads to bed erosion or incision of alluvial channels. The increase in channel gradient within the straightened reach initiates erosion in the upstream reaches. The erosion will migrate further upstream in the form of knickpoints (Brookes & Gregory 1988; Simon 1989; Shields *et al.* 1995a). The process of upstream migrating erosion knickpoints was evident in the Kuils River in 2001. No bank erosion occurred during 2002 because of the channelisation, but bank erosion occurred upstream of the channelised Site 1, undercutting bridge pillars of the Old Bottelary Road bridge (Figure 13.4). The Bunyip River in Australia was also straightened to drain the Koo-Wee-Rup swamp. This resulted in the upstream migration of a zone of bed incision along the main river and tributaries, eroding bed and banks and causing roads to collapse (Brizga *et al*. 1999b).

Channelisation also changes channel morphology depending on whether the channel was straightened, widened, narrowed or deepened. It results in straight, uniform channels (Petersen *et al.* 1992) that, in the Kuils River resulted in a straight uniform channel with low substratum and flow diversity.

The banks of the Kuils River consist of sandy non-cohesive soils and non-woody vegetation with shallow root systems that did not provide sufficient stability during the high rainfall events of May and July 2001 (Figure 13.3). These factors contributed to the mass bank failure, but were not solely responsible for it. The bank collapse was probably a result of the high rainfall, the poor stability of the banks due to their soils and vegetation, and the channelisation of downstream reaches.

13.5.2 Macro-invertebrates

There was a clear separation between biotopes in Site 1 with a low similarity between the groups. Group 1 consisted of invertebrates that lived in vegetation and Group 2 those that lived in sand.

Group 1 consisted of snails (Physidae and Lymnaeidae), dragonflies (Gomphidae), damselflies (Coenogrionidae), beetles or water bugs (Dytiscidae) and mayflies (Baetidae). Group 2 consisted of worms (Naididae) and non-biting midges (Chironomidae). Group 2 could not live in vegetation because they were not adapted to live in those conditions, and had low densities, possibly due to shifting substrata, siltation and the absence of aquatic vegetation (Junk *et al.* 1989). There was no clear correlation between groups and biotopes, flow types or year.

It was expected that biodiversity would decrease downstream with Site 4 having the lowest diversity and Site 1 the highest. This was not the case. Sites 1 and 4 had almost the same number of families (seven and five respectively) for 2001. Sites 2 and 3 had the highest diversity with 11 and 14 families respectively for 2001 (Table 13.7 and 13.8 and Appendix 13.1). The higher diversity in sites 2 and 3 could be attributed to the higher biotope diversity (Table 13.4).

The group compositions decreased during 2002 except for site 4 (Table 13.7 and 13.8), perhaps due to the loss in the variety of biotopes in 2002 with channelisation. Such a loss in variety of biotopes and consequent reduction in species diversity occurred in the Danube River, Austria (Tockner & Schiemer 1997) and in rivers in agricultural areas in Finland, Sweden, Denmark and the USA (Petersen *et al.* 1992). Sediments carried down by floods could be responsible for the decrease in available habitat and consequent decrease in community composition in Site 3 during the 2002. Shieh *et al.* (1999) studied the relationship of macro-invertebrate assemblages to water quality in a river impacted by agricultural activities, dams and municipal and industrial wastes and found that changes in physical habitats were the most important factor affecting the assemblages.

Drift is the process whereby invertebrates re-colonize disturbed areas by drifting from upstream reaches. Channelisation and flooding in the Kuils River caused disturbances in the macro-invertebrate assemblages in Sites 1, 2 and 3. The lower abundances during 2002 might have been because samples were taken too soon after these disturbances, when suitable habitats were not available. Invertebrate drift from upstream reaches could also explain the higher abundance and community composition in site 4 during 2002. This was not a particularly suitable environment, but it could have been receiving animals drifting from unsuitable areas upstream.

The families found in the study were robust. There was a good representation of predators (Gomphidae, Coenogrionidae, Aeshnidae and Libelullidae), worms (Naididae) and non-biting midges animals (Chironomidae), as well as those that lived in the water column (Dytiscidae, Corixidae, Gerridae, Veliidae and Culicidae) and snails (Physidae and Lymnaeidae). The worm and wormlike animals could live in the sand where they fed on organic matter in the sand. The presence of snails was indicative of water rich in calcium, which is necessary for their shells. The families were typical of lowland rivers in summer (Davies & Day 1998), while high abundances of Oligochaeta, Nematoda, Simuliidae and Chironomidae are characteristically recorded in rivers impacted by industrial and municipal waste in urban areas (Shieh *et al*. 1999). Winter & Duthie (1998) also found higher abundances of Oligochaeta and Simuliidae in urban rivers than in rural rivers.

13.5.3 Vegetation

There was a clear distinction between those plant species found on top of the bank and those found in wetter conditions in Site 1 during 2001 (Figure 13.13 and 13.14). This was not the case in 2002.

Figure 13.14 also showed that that there was no clear pattern with respect to plant assemblage position on the cross-section in 2002. Group 4, for example, occurred in different places along the profile. The groups identified were different to those found in 2001, and contained many different species. Weeds occurred in both years (Appendix 13.2). Many would have been introduced through agriculture and early settlement, with no natural predators to keep them within bounds. They can reproduce rigorously and rapidly due to their ability to produce large quantities of seeds (Lamp & Collet 1983), and make aggressive invaders and colonisers of disturbed land.

Such land is continuously being created by droughts, flooding, trampling and grazing (Grime 1979). Channelisation in the Kuils River during October and November 2001 involved taking sediment deposited in the channel, which originated from the collapsed banks, and in-filling the same banks. The soil was disturbed during the construction phase unearthing seeds in the seed bank, bringing them closer to the soil surface and exposing them to light. The banks were also left bare with only a thin strip of *Pennisetum clandestinum* planted near the water surface. The banks remained bare until January (normally the driest time of the year) 2002 when the Western Cape received unexpected rainfall. The moisture input and readily available seeds beneath the soil surface resulted in the rapid germination of annual, alien weeds.

13.6 Conclusions – Impacts of channelisation

The Kuils River provided a classic case of the impacts of channelisation on geomorphology and ecology. The following are summarised points on the impacts. The results of this study support those reported in the international literature.

Impacts on geomorphology

- Channel widening and straightening resulted in a uniform channel shape.
- This reduced substratum and flow diversity.
- Channelisation was followed by an upstream migration of bank erosion, which in turn, was followed by massive bank collapse during the following wet season.

Impacts on ecology

- The uniform channel shape and reduced substratum and flow diversities resulted in a reduced diversity of aquatic habitats.
- This correlated with a reduced diversity of aquatic invertebrates.
- Channelisation and bank re-construction disturbed the soil and the vegetation.
- This resulted in a higher diversity of plant species the following year, most of which were alien weeds.

Impacts on infrastructure

- Following channelisation, the upstream progression of bank erosion led eventually to undercutting of the Old Bottelary Road bridge pillars.

14. THE SILVERMINE RIVER

Karl Reinecke and Jackie King, Freshwater Research Unit, University of Cape Town

14.1 Introduction

This chapter is a summary of a three-year research project. Further details will be provided in the MSc thesis of Karl Reinecke.

14.1.1 Description of the catchment

The Silvermine River rises at an altitude of 640 m in the Steenberg Mountains at 34^{o}07'S and 18^{o}27'E, 10 km South of Cape Town (Figure 14.1). The river derives its name from attempts, in the late 17th century, to extract silver from its catchment (Cobern 1984). It is a short, naturally perennial river approximately 12 km long, with a catchment area of *ca*. 21 km^2.

The Silvermine catchment is in the Winter Rainfall Region, and the river has a mean annual runoff of 4.5x 10^6 m^3 a^{-1} at the reservoir in the upper catchment (Heinecken 1982). Highest rainfalls typically occur from June to August, with February being the driest month. Rainfall in the upper catchment is in the order of 1200 mm a^{-1}, whilst the lower slopes are much drier with Fish Hoek receiving a mean of 600 mm a^{-1}.

The upper catchment consists mainly of quartzitic sandstones of the Table Mountain Group, which overlay shales of the Malmesbury Series. These are highly leached and acidic in the rocky outcrops (Heinecken 1982). There is a high-altitude wetland in the upper catchment with organically rich colluvial sands (Harding 2000). In the middle catchment, the lower mountain slopes consist of wind-blown (Aeolian) sands and shelly material, which overlay weathered Cape Granite (Heinecken 1982). These were blown up the valley from the lower-lying sand dunes, forming a deeper mixed soil. The lower catchment has marshes bordering the river, with organically rich, sandy materials (Harding 2000).

There is a small dam in the upper reaches, just downstream of the high-altitude wetland, which was constructed in the late 19th century to provide water for Kalk Bay and Muizenberg. The reservoir has a capacity of 83,000 m^3. Water soon begins to flow over the spillway after the winter rains start, but little moves past the dam and down the river in summer. One kilometre downstream from the dam is a weir that diverts stream flow into a pipe, along with water that has been piped underground from the reservoir to this point. The water is sent along the old Kalk Bay/Muizenberg pipeline, with varying amounts used by Westlake Golf Course (WLGC) and the remainder emptying into Sandvlei coastal lake. Although some stream flow moves past the weir offtake in winter, none does in summer, and so there is no water in the channel downstream of the weir in summer until the first tributary (flow site 6; Figure 14.2) joins the main channel just upstream of Ou Kaapse Weg. There are five tributaries, three joining the left bank and two the right bank. Tributary 5, the most downstream one, is only known from maps; the channel, if it still exists, was not found. Several parts of the drainage system are dry in summer or consist of stagnant water (Table 14.1).

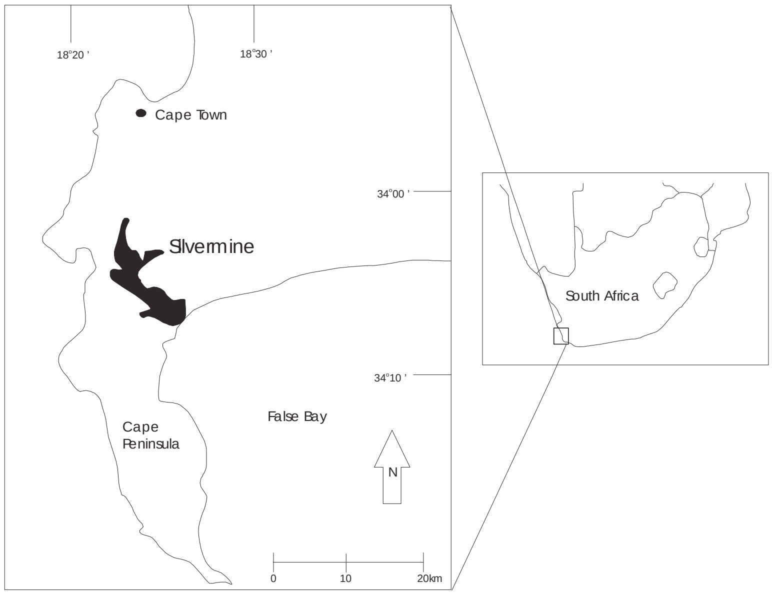


Figure 14.1 Location of the Silvermine section of the Cape Peninsula National Park.

An access track used to surround the high-altitude wetland (Figure 14.2). Numerous seeps draining toward the track were diverted into storm-water drains, and these changed the flow pattern into the wetland from general seepage to small channels that have eroded the peaty soils. Use of the track was recently discontinued to allow the wetland vegetation to re-grow.

Table 14.1 Perenniality of flow sites in summer. NPF = no perceptible flow, i.e. where measured water velocity <0.001 m s^{-1}. Flow sites as per Figure 14.2.

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

Several houses have been built in the coastal floodplain and a few actually incorporate the river channel into their gardens. Residents have boreholes that tap the aquifer of the floodplain. The Clovelly Country Club (a golf course), situated adjacent to the same floodplain, also uses the river water extensively for irrigation. There is some indication that the river course may have been changed and restricted in order to increase the extent of the golf course fairways and greens. Urban areas encroaching on the coastal floodplain are prone to flooding during peak rains. In order to combat this, three artificial wetlands have been built or are planned for the area just upstream of where the coastal road crosses the river.

There are no serious problems of water pollution in this catchment, due to the lack of urban, agricultural and industrial areas. Traffic along Ou Kaapse Weg produces litter that drains toward the river, although there is little evidence of it in the channel. The semi-urban areas in the lower reaches must have some impact on the river, but the extent of this is not known (Harding 2000).

Until 1998, the upper catchment and the higher parts below the road were owned by Cape Town City Council, known as Silvermine Nature Reserve, and used extensively for public recreation. There were picnic spots along the river, among plantations of *Pinus radiata* (pine). The middle catchment was farmed with crops and livestock. The valley slopes of the middle catchment are still largely covered with *Pennisetum clandestinum* (kikuyu grass) on man-made terraces, and dense stands of alien *Acacia* persisted after cessation of the farming practices in 1968 (Cape Peninsula National Park pers. comm.).

Both the upper and middle catchments were incorporated into the new Cape Peninsula National Park (CPNP) in May 1998, and the old homestead became the peninsula headquarters for National Parks personnel. The lower catchment encompasses the urban areas and the golf course.

14.1.2 Recent disturbances: fire and clearing of alien vegetation

This WRC project was closely linked to a number of disturbances, planned and unplanned, that affected the river during the project's life (Table 14.2). Extensive mountain fires destroyed much of the catchment's vegetation in January 2000, with all areas upstream of Main Study Site 6 burnt (Figure 14.2). Areas previously covered by thick *Acacia* and pine forest were laid bare. After the fires, new fire policies were implemented by CPNP that resulted in eradication of the remaining alien vegetation. Adult *P. radiata* upstream of Ou Kaapse Weg that were not burnt were cleared towards the end of 2000, with follow-up clearing of germinating saplings completed by members of the public in conjunction with the CPNP personnel. Roots and short stumps of the old trees remain visible, but few, if any, pine saplings were present as at March 2003.

Clearing *Acacia* from the middle catchment by Working for Water (WfW) teams presented a more difficult problem. The main species present are *Acacia longifolia* (long-leafed wattle) and *Acacia saligna* (port jackson willow). Vigorous re-growth of *Acacia* seedlings after the fire has continued in this area despite sporadic clearing activities from late 2000 to date. A further area of concern was that surrounding the river adjacent to and upstream of the homestead. *Populus canescens* (grey poplar) had formed an almost impenetrable stand along, and partially within, the river channel. CPNP personnel cleared this to some extent during 2001/2, but a large number of adult poplars remained as at March 2003. *P. clandestinum* covers most of the area around the homestead.

Other than clearing of alien vegetation, CPNP policy is not to undertake active river rehabilitation, but to rely on natural rehabilitation processes. This has created the opportunity to study responses of the river, as much of its catchment develops a new cover of native vegetation whilst other parts are still subject to clearing activities.

Table 14.2 Timeline for duration of project showing relevant disturbances of the catchment, and field work.

Event	2000				2001				2002			
	J-M	A-J	J-S	O-D	J-M	A-J	J-S	O-D	J-M	A-J	J-S	O-D
Fires cleared almost all vegetation upstream of Main Study Site 6	X											
Remaining pines above Ou Kaapse Weg cleared		X	X	X								
First summer of field work completed					X							
Acacia and *P. cansescens* cleared from middle catchment				X	X	X	X	X	X	X	X	X
Second summer of field work completed									X			

14.2 Project aims and objectives

14.2.1 Objective

The overall objective of the Silvermine research project was to record the natural rehabilitation of the river after the January 2000 fires and catchment disturbance associated with clearance of alien vegetation.

14.2.2 Aims

The aims were to record:

- characteristics of the macro- and active channel, and to track changes in these and aquatic habitat from one summer period to the next, due to winter floods and instability in the riparian zone;
- the nature and re-growth of the riparian and aquatic vegetation through one year of seasonal changes and alien clearance;
- aquatic invertebrate fauna twice, at yearly intervals, as indicators of River Health;
- the present hydrological regime, to the extent possible, with spot measurements of discharge;
- water quality over one year.

14.3 Materials and methods

14.3.1 Study sites

Seven study sites were chosen. Four Main Study Sites (Sites 2, 4, 5, and 6) formed the focus of the project and the remaining three Supplementary Sites (Sites 1, 3 and 7) served to describe channel geometry other than that at the main sites. The data collected at each of the seven study sites are summarised in Table 14.3.

The study sites were chosen to represent a diversity of alien vegetation and to span the upper, middle and lower catchment (Figure 14.3). The sites differed in the severity of invasion and species of alien vegetation present. A summary of all the study sites, their locations in the catchment and other key features used in assigning zonation down the river are given in Table 14.4. X co-ordinates are distances in metres from the equator. Y co-ordinates are distances west of the 23^{o} line of longitude (which is the location of Cape Town). Features used to assign zonation were taken from King & Schael (2001). A more detailed description of the seven study sites follows, with details of channel maps and surveyed cross-sections provided in Appendices 14.1 and 14.2. Right bank is the right bank when facing downstream.

Table 14.3 Data collected at four Main and three Supplementary Study Sites

Data collected	Sites 1,3,7 (supplementary)	Sites 2,4,5,6 (main)
Surveyed cross-sections	X	X
Vegetation transects		X
Aquatic macro-invertebrates		X
Aquatic habitat maps		X
Discharge readings	X	X
Water quality (pH, T^{o}, conductivity)	X	X

Site 1 (Supplementary Site)

A single cross-section was surveyed in the source zone upstream of the reservoir. This upland wetland basin is ca. 400 m in width and characterised by spongy, peaty soils. The area was previously afforested with *P. radiata*, most of which were burnt in the January 2000 fire. The remaining few were felled and removed within a year of the fire, as were new individuals. The channel was less than 1 m wide and flow was perennial. The substratum consisted of large and small cobbles in sand.

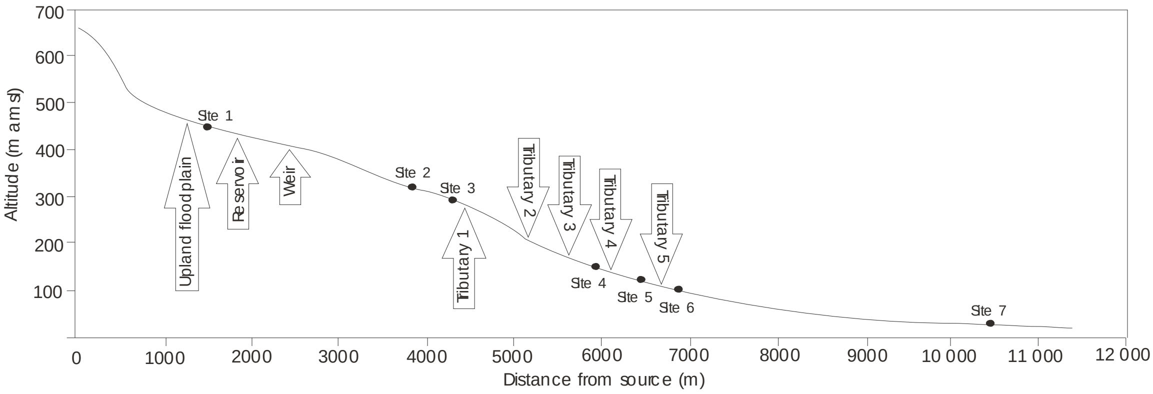


Figure 14.3 Longitudinal profile of the Silvermine River. The seven study sites are shown in relation to key features illustrated in Figure 14.2.

Site 2 (Main Site)

Site 2 was situated downstream of the weir and upstream of Ou Kaapse Weg in the mountain-stream zone of the upper catchment. The site was approximately 50 m long by 60 m wide. The area was in a pine plantation prior to the January 2000 fire. The remaining standing pines were felled and removed within a year of the fire, and new individuals were routinely cleared. The channel was about 3 m wide and fringed by adult *Cunonia capensis* and *Virgilia oroboides*, interspersed with *Todea barbara* and *Prionium serratum*. The left bank sloped gently and, during the study, supported young growth dominated by *Psoralea pinnata* and *V. oroboides*. The right bank was a largely un-used picnic site with many alien trees, which remained standing pending plans to re-landscape the area. The channel substratum was mixed bedrock and boulders, interspersed with cobble, gravel and sand. The channel outline was surveyed and three cross-sections established.

Table 14.4 **Location and key features of the seven study sites.** Co-ordinates given are those of the Right Bank control of the first cross-section at each Main Study Site. They are presented in the Gauss Conform Projection based on the Clarke 1880 elipsoid. Co-ordinates were not recorded for the Supplementary Study Sites (#).

Site #	Co-ordinates Y	Co-ordinates X	Zone	Alien species present	Slope	Altitude (m asl)	Distance from source (m)
1	#	#	Source zone	*P. radiata*	0.044	460	1450
2	53639.5	3773622.3	Mountain-stream	*P. radiata*	0.057	320	3750
3	#	#	Mountain-stream	*P. radiata*	0.074	280	4200
4	53421.6	3775361.8	Foothill	*Acacia*	0.049	140	5975
5	54165.9	3775430.1	Foothill	*Acacia*	0.037	115	6525
6	54420.1	3775663.6	Transitional	*P. canescens*	0.028	100	7250
7	#	#	Lowland	*Acacia*	0.008	20	11350

Site 3 (Supplementary Site)

Site 3 was some 550 m downstream of site 2 and also upstream of Ou Kaapse Weg. A single cross-section was established. The channel was surrounded by dense, young *V. oroboides* thicket, situated on steep rocky banks.

Site 4 (Main Site)

Site 4 was located just upstream of tributary 4. This was the largest site, being 100 m long and 80 m wide. It was chosen because of the high degree of mobility of the channel, with high, almost vertical, sand banks edging the river. The area was heavily infested with alien *Acacia* before the fire, and vigorous re-growth continued despite repeat clearing by WfW and CPNP staff. The macro-channel was last cleared of aliens in July 2001. Heaps of hacked vegetation were stacked on the banks, within the macro-channel, and large logs and other debris were strewn across and in the active channel. This site had the fewest number of indigenous plant species and the greatest coverage of alien vegetation species. The channel was approximately 5 m wide, with a mixed substratum ranging from bedrock and boulders to gravel and sand. The channel outline was surveyed, and five cross-sections established.

Site 5 (Main Site)

A few hundreds of metres downstream, in an area with lower banks, Site 5 was half the size of Site 4 (50 m x 30 m). After the January 2000 clearance by fire, *Acacia* re-grew less vigorously than at Site 4 and were largely cleared from the area in July 2001. There are a greater variety and cover of indigenous vegetation than at Site 4, including *Juncus capensis*, *J. effusus*, *Pycreus polystachyos, Carpha glomerata*, *Lauremburgia repens* and *Isolepis prolifer* at the channel margins. Further from the channel *Pteridium aqualinum* and *Senecio rigidus* dominate. The channel was 5 m wide with bedrock and boulders separating shallow sandy pools. The channel outline was surveyed and three cross-sections established.

Site 6 (Main Site)

Site 6 was situated in the poplar forest, just upstream of the homestead and about 700 m downstream of site 5. The dense forest had an understory of *Lantana camara*, making it almost impenetrable. Due to this and the subsequent partial clearance of the vegetation by CPNP staff, there was a large amount of woody debris in the channel throughout this project. Some of this formed debris dams that trapped sediments washing downstream. Stunted *Kiggelaria africana* and *Chionanthus foveolatus*, revealed during the felling operations, were two of the very few native woody species in the area. The channel was also about 5 m wide and had a mostly sandy substratum with scattered cobble and gravel. The channel outline was surveyed and three cross-sections established.

14.3.2 Surveying channel outlines and cross-channel profiles

Channel outlines were surveyed to produce longitudinal maps of the sites (Appendix 14.2). Surveying was done by graduate students at the Geomatics Department at the University of Cape Town, using an electronic theodolite (Leica TC307 model) and a standard Leica prism on a staff (Table 14.5). Specifically, MC Briers and P Bornman completed the required surveying in the summer and winter of 2001 and P Bornman alone in the summer of 2002.

Seventeen cross-sections were also surveyed in across the macro-channel, at points deemed representative of different conditions at the sites (Appendix 14.1). The cross-sections are numbered by site, and in sequence within a site from upstream to downstream (e.g. CS 6-2 is the second most upstream cross-section at Main Study Site 6). Each cross-section extended through the active channel and beyond the macro-channel on both sides in a straight line, with a permanent cemented beacon at each end. The cross-sections were used to record the bank-to-bank channel shape; re-surveying of them at later dates allowed any changes in channel shape to be recorded (Table 14.5). All the beacons were surveyed to a single master beacon, and to the nearest trigonometric beacon. The absolute heights (metres above sea level) and distances across each of the surveyed cross-sections are summarised in Appendix 14.3. The surveyors used the Computer Aided Drafting (CAD) programmes Autocad and Allycad to generate the maps and cross-sectional profiles.

Table 14.5 Survey dates and purpose.

Survey	4 Main Sites	3 Supplementary Sites
Cross-sections	30 Nov–15 Dec 2000	30 Nov–15 Dec 2000
Channel outline	30 Nov–15 Dec 2000	30 Nov–15 Dec 2000
Flood-water levels	25 Jul, 1-8 Aug, 21 Sept 2001	
Cross-sections	24-25 Jan 2002	24-25 Jan 2002
Channel outline	24-25 Jan 2002	

14.3.3 Vegetation transects

Three transects were established at each of the four main sites, in order to record distribution, density and condition of all vegetation species. The transects were positioned on existing surveyed cross-sections (Table 14.6). Data were collected from back-to-back, metre-square, groundcover vegetation plots along the complete length of each transect in the summer of 2001 and again the following summer. In this way, a total of 576 m^2 plots were sampled each year. Each plant in each plot was identified to species, and the areal coverage of each species in each plot was recorded as a percentage, together with the greatest height per species and average condition (dead, half-dead, alive) per species. A specimen of each species was collected from outside the plots, and pressed for later identification by specialists. Species that were flowering or setting seed were noted. Five measurements of soil depth were made in each metre-square plot by hammering a rod into the ground just inside each corner and then once in the middle. The soil data are not presented in this summary.

Table 14.6 Vegetation transects sampled at each Main Study Site, their relation to surveyed cross-sections, and the number of species recorded in 2001 and 2002. The transect number is the same as the cross-section on which it was positioned.

Site	Transect number	Transect length (m) and number of plots	# species 2001	# species 2002	# species % change
2	2-1	61	36	42	17
	2-2	60	29	52	79
	2-3	51	37	48	30
4	4-1	49	38	35	-8
	4-3	78	29	32	10
	4-5	98	43	38	-12
5	5-1	22	22	25	14
	5-2	24	25	24	-4
	5-3	25	28	23	-18
6	6-1	34	7	27	286
	6-2	22	9	13	44
	6-3	53	8	17	113
Total	**12**	**577**			

Site 6 also contained dense stands of trees, the distribution of which could not be recorded with the small plots. Six one hundred metre-square tree plots were therefore used to record tree distributions, using the same variables as for the small plots. Four of the large plots were at cross-section 6-3, two at cross-section 6-2, and one at cross-section 6-1.

All vegetation data were entered into the vegetation database TURBOVEG V1.99h (Hennekens 1996). TURBOVEG outputs data files that are compatible with most common analysis packages such as Microsoft EXCEL, CANOCO (ter Braak & Smilauer 1998) and PRIMER (Clarke & Gorley 2001), with PRIMER used in this analysis. Fieldwork was done under the guidance of Project Advisor Dr Charlie Boucher of the University of Stellenbosch.

14.3.4 Aquatic habitat maps

Maps were drawn of the aquatic habitat at all four main study sites in the summers of 2001 and 2002, in order to record habitats over a span of one year and assess any change. The concept of, and background to, hydraulic biotopes were described by Rowntree & Wadeson (1999). The variables used in this present study to describe habitat were similar to those used by King & Schael (2001) to define their hydraulic biotopes. They used two visual criteria: flow type and the underlying substratum. In their definition of hydraulic biotopes, eight categories of substratum and 14 categories of flow types were used. These were reduced to six categories of each of flow and substratum types, under the guidance of the project leader Dr Jackie King. A further four habitat variables were also used to help differentiate habitat types (Table 14.7).

Table 14.7 **Habitat descriptor codes.** Substrata classified according to the Wentworth scale (Rowntree & Wadeson 1999).

Code	Substratum Type	Code	Flow Type	Code	Other
SI	silt	ST	still	MV	marginal vegetation
SA	sand	BF	barely flowing	IV	in-channel vegetation
GR	gravel	SS	slow smooth	OR	organic litter
CO	cobble	MR	medium rippled	AL	algae
BO	boulder	TR	trickle		
BR	bedrock	CA	cascade		

The previously surveyed channel outlines were used as templates for all the hand-drawn habitat maps. A tape measure was laid between the marked cross-sections, to guide mapping of the distributions of different-sized substrata. Three overlays were then drawn: of flow types, marginal vegetation and canopy cover. Because flow types change with flow volume, discharge was recorded on the same day that a flow map was drawn. The maps were scanned in and imported to ARCVIEW 2.3a for digitising and calculations of the proportions of different habitats. Digitising was done by A. Plos and P. Pillay of the Zoology Department, UCT. The digitised flow and substratum maps for Main Study Sites 2 and 5 are shown in Appendix 14.4, and for Main Study Site 4 in Section 14.4.4.

14.3.5 Aquatic macro-invertebrates

Twenty-nine invertebrate samples were collected during each of the two summer sampling periods. At each site all available combinations of the six substratum and six flow types were identified, and one sample taken from each recognised combination (habitat). Qualitative rather than quantitative samples were taken due to the large range in size and shape of the identified habitat types (Table 14.8). It is thus not possible to compare absolute abundances of invertebrates per habitat, only the proportions of different taxa. Specimens were collected with a rectangular hand net 0.20 m x 0.15 m with a 250 μm mesh size. A soft plastic hand brush was used to scour bedrock and to reach between boulders. Gravels were disturbed by hand, whilst silt and sand were collected and later searched for animals through flotation and sieving. Specimens were preserved in 70% alcohol.

All macro-invertebrate data were analysed for distribution patterns using the multivariate statistical package PRIMER 5. Data were stored in EXCEL.

14.3.6 Discharge readings

This additional activity, partially funded by the City of Cape Town, was done to map the main contributing areas of surface flow through the catchment at monthly intervals over one year. In the absence of a comprehensive history of river flow in this catchment, the knowledge was gathered as a contribution to future discussions between management bodies and water users on flow management of the river.

Fifteen flow sites were chosen in the catchment, at which discharge readings were taken regularly (Figure 14.2). Seven of these were the study sites and the other eight were located at key points, such as on important tributaries. Six of the flow sites were in the upper catchment and the remaining nine in the middle and lower catchment. Four were on the most important tributaries, all of which are unnamed. A fifth tributary indicated on a National Parks office map was not found. Each month, discharge was measured at every site on the same day. Readings were taken at least four days after major rainfall events, in order to allow the system to return to fairly stable low flows.

Table 14.8 The number of invertebrate samples per site in 2001 and 2002, and the kind and area of substratum-flow combinations sampled. Acronyms as per Table 14.7. Data collected in April 2001 and March 2002.

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

On each monthly occasion at each of the 15 flow sites, measurements were taken at three different cross-sections. Mean column velocity and water depth were measured at eight to 15 points across each of the three cross-sections. Velocity was measured using a Marsh McBirney Electronic Flo-Mate current meter, on a top-setting wading rod. Discharge was calculated using the velocity-area method described by Gordon *et al.* (1992). The discharge values calculated for the three cross-sections at each site were averaged to provide a single value for that flow site on that day. Fifteen sets of readings were taken between April 2001 and February 2003.

14.3.7 Measurements of water quality

No in-depth analyses of water quality were done, but field measurements of water temperature (glass alcohol thermometer), and conductivity and pH (CRISON pH/MV-506 and CDTM-523 hand held instruments respectively), were made on each field visit.

14.3.8 Summary of data collected

In summary, a range of variables was measured and samples taken over two summers and one winter, as per Table 14.9.

Table 14.9 Summary of collected data, format for data storage and analytical programs used.

Category	Status
Surveyed channel outline	
Number	One for each Main Study Site for each year
Data	Stored in ALLYCAD 2.5
Analysis	Template for habitat studies
Surveyed cross-sections	
Number	17 cross-sections over 7 sites surveyed once per year for two years
Data	Stored in ALLYCAD 2.5
Analysis	Support data for vegetation transects
Vegetation transects	
Number	12 transects at four Main Study Sites totalling 577 relevés per year for two years
Specimens	Species pressed, dried, catalogued and identified
Data	Species data stored in TURBOVEG 1.99h
Analysis	PRIMER 5
Aquatic macro-invertebrates	
Number	58 samples at four Main Study Sites: 29 in 2001 and 29 in 2002
Specimens	Sorted to family and stored in alcohol
Data	Identifications on spreadsheets stored in EXCEL 2000
Analysis	PRIMER 5
Aquatic habitat maps	
Number	4 maps (base map + three overlays) per four Main Study Sites per year
Specimens	Substratum, flow type, marginal vegetation and canopy cover
Data	Field sheets digitised in ARCVIEW 2.3a
Analysis	ARCVIEW 2.3a
Discharge readings	
Number	3 cross-sections at 15 sites each measured monthly for 15 months (total of 675)
Data	Discharge data stored in EXCEL 2000

14.4 Results

In this summary chapter, one example from each of the categories of data (as listed in Table 14.9) is shown. Complete analyses and full findings will be in the M.Sc. thesis of K. Reinecke (UCT).

Site 5 is used to illustrate the results of the survey and vegetation studies, because it showed the clearest pattern of distribution of vegetation communities. Site 4 is used to illustrate the results of studies on aquatic habitat and macro-invertebrates, because it had the most mobile channel form. Discharge information is provided for flow Site 15, the most downstream one, as this is near to a recently installed gauging weir.

14.4.1 Survey and vegetation data at Main Study Site 5

The surveyed channel outline of each Main Study Site provided a basic description of the site (Figure 14.4). The channel was slightly sinuous and quite stable. Very little change occurred in channel shape over the year (Figure 14.5), despite a winter of very heavy rains. This is not the case for all the sites, with Main Study Site 4, in particular, showing considerable response to the winter floods (Appendix 14.1c, cross-section 4-4). This is discussed further in section 14.4.3.

Twenty-four vegetation relevés were studied each summer at Main Site 5, as the channel remained much the same shape (Tables 14.10-14.11). Some species were confined to the active (wetted) channel, and some to its riparian edges. Higher up the slopes the riparian species gave way to more terrestrial species. The dominant alien plants at Main Site 5 were the *A. saligna*, *A. longifolia* and *Briza maxima* (big quaking grass). In the second summer, there were fewer *A. longifolia* and *A. saligna*, and a lower proportion of aquatic plants such as *Isolepis prolifer*. Overall, this transect was quite stable both in species composition and bank shape and structure.

A summary of the species occurring at each Main Study Site is given in Appendix 14.5.

14.4.2 Multivariate analysis of vegetation data

The vegetation data were analysed using the Bray Curtis similarity and MDS ordination routines in PRIMER 5. These revealed patterns in the distribution of the vegetation assemblages (Figures 14.6 & 14.7). Seven groupings formed in the CLUSTER analysis. The same groups were repeated in the MDS ordination, indicating that they are fairly robust.

The Bray-Curtis dendrogram (Figure 14.6) indicates a first separation between the left and right banks, with the relevés from the active (wetted) channel grouping with the left bank relevés. The next level of separation is between years and height up slope, with the right bank showing four distinct sub-groups of relevés: macro-channel (i.e. highest up the slope) in 2001 and 2002, and riparian in 2001 and 2002. Macro-channel relevés from the left bank also form two groups for the two years, whilst the channel and left-bank riparian relevés form one compact group. Similar patterns showed in dendrograms for all transects at this site, with the channel relevés often grouped with the riparian relevés. Another repeated pattern, less evident in this example, was that separation of the macro-channel and riparian relevés was less obvious in 2001 than in 2002. This suggests that one year after

the fire there was poor vertical zonation of vegetal communities up the macro-channel bank, but that this was emerging by the second year.

MDS ordination (Figure 14.7) shows a similar pattern. All macro-channel relevés are on the right side of the plot, whilst the riparian and channel ones are on the left. This coincides with a gradient from drier soils on the right to wetter ones on the left. Additionally, most of the drier relevés contain alien plants, whilst the wetter ones do not.

14.4.3 Aquatic habitat maps for Main Study Site 4

At Site 4, considerable channel changes took place during the winter between the two summer sampling periods (Appendix 14.2b). A meander cut-off occurred on the left bank during floods, and a new channel was scoured. The old channel thereafter flowed only during winter. Despite the considerable erosion that occurred, with high sandy banks collapsing into the river, the proportions of different sized substrata remained stable (Table 14.12), possibly because erosion was ongoing at that site anyway.

Table 14.12 The percentages, by area, of the substratum types at Main Sites 2, 4 and 5 in summer 2001 and 2002. In 2002, the meander cut-off exposed organic soils at Main Study Site 4 totalling 2 % of the substratum (#). At Main Study Site 2 in 2002, 6% of the substratum was covered by *Todea barbara* ($).

Substratum Type	Percentage by Area					
	Site 2		Site 4		Site 5	
	2001	2002	2001	2002	2001	2002
Bedrock	19	29	3	8	8	8
Boulder	11	27	23	26	30	45
Cobble	20	20	15	15	20	7
Gravel	14	4	10	2	4	9
Sand	33	14	47	43	20	29
Silt	3	14	2	4	18	0
Total	100	$94	100	#98	100	100

Four kinds of habitat maps were used to record and describe the status of the aquatic habitats at the site: substratum type, flow type, marginal vegetation and canopy cover. The first two kinds of maps are addressed here. The main change in the distribution of substratum types was due to the new channel created by the winter floods (Figure 14.8). Downstream of the meander cut-off area, the riverbed had been dominated by sand before the 2001 floods, and after the floods domination by sand and silt throughout the cut-off area and downstream was even more extensive. Gravel areas virtually disappeared, patches of cobble and boulder were rare and isolated, and in the area upstream of the cut-off much bedrock was exposed, perhaps as finer material was stripped from the bed by faster flows.

In the following summer of 2002, more slow flow-types were mapped at Main Sites 2, 4 and 5 than in 2001 (Table 14.13), probably due to lower discharges on mapping day (Figure 14.9) and a slightly wider channel (Appendix 14.2b). Cascades were missing from the 2002 map and there was less longitudinal connection of flow types. There was no flow through the old channel, with all now moving patchily through the new cut-off channel.

Table 14.13 The percentages, by area, of the flow types at Main Sites Site 2, 4 and 5 in summer 2001 and 2002.

Flow Type	Percentage by Area					
	Site 2		Site 4		Site 5	
	2001	2002	2001	2002	2001	2002
Still	0	99	0	4	0	12
Barely flowing	1	0	37	55	46	38
Slow smooth	94	0	35	18	36	30
Medium rippled	0	1	27	23	17	20
Fast turbulent	0	0	0	0	0	0
Trickle	5	0	0	0	0	0
Cascade	0	0	1	0	1	0
Total	100	100	100	100	100	100

14.4.4 Multivariate analysis of aquatic macro-invertebrate data

The PRIMER modules used for vegetation were also used to detect patterns in the distribution of macro-invertebrate assemblages. Significant changes in aquatic habitats over the study period could be expected to manifest as changes in the presence or absence of certain invertebrate assemblages. Descriptions, and the number, of sampled habitats for each site are given in Table 14.8. Bray-Curtis similarity analysis of family-level invertebrate data (Figure 14.10) revealed six groups of samples. The groups were to some extent separated by habitat, and to some extent by year, but the pattern was poorly defined. The 2002 ‘year group’ was somewhat better defined into sub-groups than the 2001 ‘year group’, with the fauna of faster flowing, rocky habitats tending to separate from those of the slower flowing, sandy habitats. Both ‘year groups’ were separate from two samples taken in marginal vegetation.

The MDS ordination (Figure 14.11) revealed the same pattern, with most samples from 2001 clustering at the bottom right of the plot and those from 2002 at the top left. Again, the distinction of rocky and sandy samples was clearer in 2002 than in 2001, although the two groupings were loose, indicating a low level of similarity in species composition. It is possible that the invertebrate assemblages were reflecting some impact of the 2000 fires, through indistinct habitats with poor heterogeneity in 2001, and with some slightly increased habitat heterogeneity and resolution of assemblages in 2002.

14.4.5 Discharge readings

Fifteen sets of discharge readings (Table 14.14) were taken from April 2001 to February 2003 at 15 flow sites (Figure 14.2). The pattern was one of high winter flows and low summer flows (Figure 14.12). Flows were higher in the first winter than in the second.

Discharge values along the river reveal the impact of the weir (Figure 14.13). In summer, the weir takes all the flow in the river (flow site 3a), leaving the next flow sites downstream dry (flow site 4) or with small standing pools (flow site 5). Flow only begins again in the mainstream (flow site 7), probably due to seepage from the pools at flow site 5 and others in the first tributary (flow site 6). In winter, the weir takes some of the flow (flow site 3a), with the remainder over-topping the weir to contribute to the overall downstream increase in discharge values.

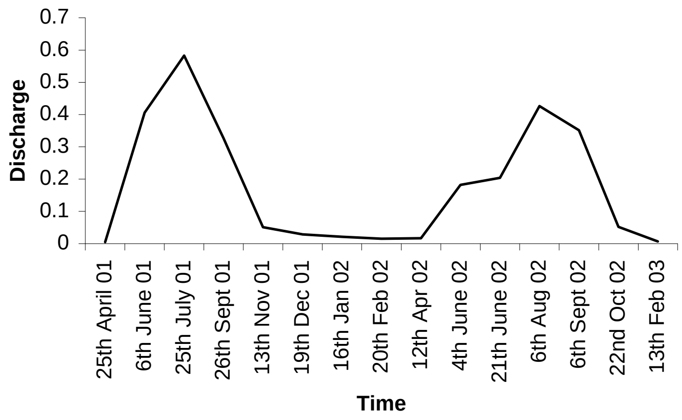
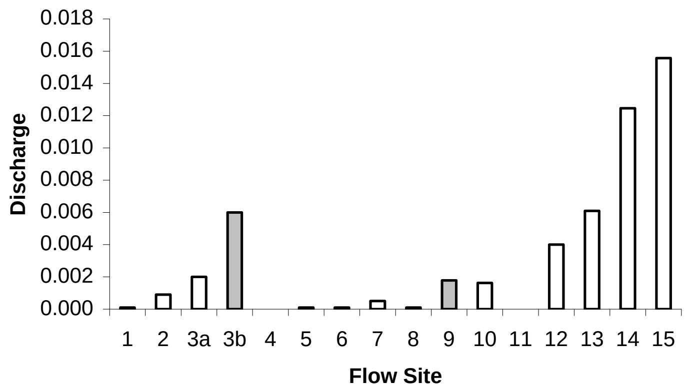


Figure 14.12 Discharge ($m^3 s^{-1}$) at monthly intervals at flow site 15

Water from the weir moves to a tank located on the Silvermine Naval Complex and above Westlake Golf Course (WLGC). From there, some flows to WLGC and the rest empties into Sandvlei (Figure 14.14).

a.



b.

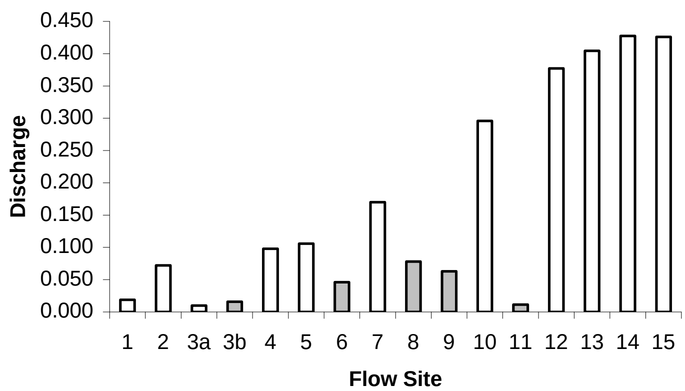


Figure 14.13 Discharge at each flow site ($m^3 s^{-1}$) in (a) a low-flow summer month (20 Feb 2002) and (b) a high-flow winter month (6 Aug 2002).

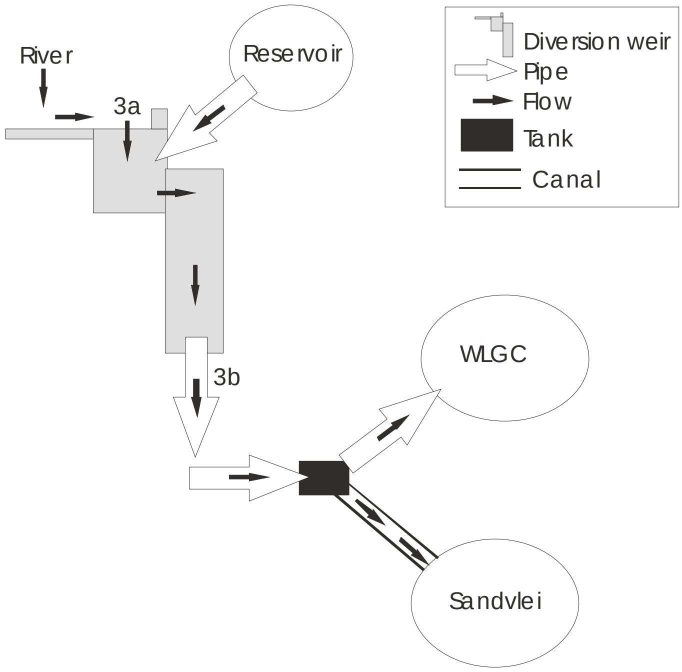


Figure 14.14 Schematic of the movement of water through the weir at flow site 3.

14.4.6 Measurements of water quality

Field measurements of water temperature, conductivity and pH (Table 14.15) were taken on all occasions when discharge was measured except when instruments were mal-functioning. The only water-quality variable found previously to have been measured is a pH reading of 4.9 at the reservoir (McVeigh cited by Heinecken 1982). In the same CSIR report on the status of the Silvermine Estuary, other water-quality readings were taken but only at the estuary. The values recorded during this project are thus compared to the water-quality guidelines for the southern and western coast presented by Dallas *et al.* (1998) and repeated here in Table 14.16.

Conductivity

The conductivity recorded in the mountain-stream flow sites (2, 4, 5 and 6) fell within the range of expected values, except during June of both years when it was higher. Usually conductivity decreases during winter (Dallas *et al*. 1998), but in this case there was no such pattern. All readings at the foothill flow sites (10 and 12) were higher than the average and the maximum reported by Dallas *et al.* (1998), but the Dallas *et al.* values are unusual in that they are lower than their mountain-stream values. The values in the transitional zone (flow site 13) are higher than the average but within the upper maximum limit, while those in the lowland (flow sites 14 and 15) are within the stated very wide range.

pH

pH values at the mountain-stream flow sites were lower than the regional average, except for one month (February 2003) which gave a questionable high reading. Values recorded at the foothill flow sites (10 and 12), the transitional zone pH (flow site 13) and the lowland (flow sites 14 and 15) were within the regional ranges, and generally lowest in winter and highly variable seasonally.

Table 14.15 Water-quality values from main channel flow sites.

Flow site	16 Jan 01	14 Jun 01	13 Nov 01	21 Jun 02	6 Aug 02	13 Feb 03
			Temperature °C			
2	18.6	11.5	22.6	10.0	12.0	18.0
4	27.1	10.5	27.5	10.0	12.0	DRY
5	21.4	11.0	23.3	10.0	11.0	19.0
7	17.1	9.0	17.1	8.5	11.0	19.0
10	19.9	11.0	18.9	9.0	12.0	25.0
12	20.4	12.0	20.9	10.0	12.0	25.0
13	20.8	12.5	21.5	10.0	13.0	24.0
14	21.5	12.5	20.8	10.0	12.0	24.0
15	22.2	125.5	21.0	10.0	12.5	24.0
			Conductivity mS m^{-1}			
2	11.9	27.3	5.8	26.8	8.4	11.8
4	18.0	32.5	12.0	31.6	10.7	DRY
5	18.0	31.2	10.8	32.4	11.3	16.3
7	19.5	36.1	14.1	37.2	10.9	31.3
10	23.4	35.9	17.1	42.7	14.9	31.2
12	25.7	40.4	19.5	41.0	15.7	27.5
13	26.9	39.6	21.0	41.9	16.0	26.0
14	55.4	47.9	32.0	48.1	18.0	57.0
15	67.5	53.4	33.3	53.7	19.7	68.3
			pH			
2	4.6	3.5	4.2	3.7	3.9	6.8
4	4.2	3.8	4.4	3.7	3.3	DRY
5	4.0	3.7	4.2	3.5	3.7	4.5
7	4.5	4.0	3.9	3.3	3.9	6.6
10	7.5	4.0	3.9	3.6	4.0	4.2
12	7.5	4.1	3.9	3.8	4.1	3.8
13	7.5	4.2	3.5	3.9	4.3	4.9
14	7.5	5.3	6.7	5.6	5.5	6.3
15	7.0	6.0	6.5	6.3	6.2	6.2

Table 14.16 Zonation of main channel flow sites, and the water-quality guidelines (Dallas *et al.* 1998) for these zones in south and west coast rivers.

Flow Site	Zone	Variable	Average ± SD	Minimum	Maximum
2,4,5,7	Mountain-stream	Conductivity (mS m^{-1})	3.6 ± 2.5	0.9	21.5
		pH	5.5 ± 0.9	3.6	7.9
10,12	Foothill	Conductivity (mS m^{-1})	3.8 ± 2.1	1.5	11.2
		pH	5.8 ± 0.8	4.0	7.2
13	Transitional	Conductivity (mS m^{-1})	10.8 ± 6.6	2.6	47.9
		pH	6.5 ± 0.8	4.3	8.9
14,15	Lowland	Conductivity (mS m^{-1})	39.0 ± 25.5	4.5	107
		pH	7.3 ± 0.4	6.5	8.5

14.5 Discussion

The overall objective of this project was to monitor and record any changes to the river and its associated habitats as recovery took place after the fires in January 2000 and as the remaining alien vegetation was cleared.

The baseline set of summer data, collected from December 2000 to March 2001, recorded conditions one year after the fire. The second summer's data recorded conditions two years after the fire, but also reflected the river's response to the follow-on clearance of alien vegetation in the catchment. As CPNP follows a non-interventionist policy other then alien clearance, any rehabilitation of the river and its catchment took place largely unassisted.

Generally, indigenous vegetation returned more quickly in the upper catchment than in the middle catchment, with the few pine saplings that appeared actively removed by a volunteer group. In the middle catchment, the *Acacia*, both on the hillslopes and in the macro-channel, hampered re-growth of indigenous plants, and WfW teams and CPNP personnel continued an active programme of alien clearance throughout the project. Clearance of areas was done as and when funds and personnel became available, without discussion with researchers from this WRC project. It was thus impossible to design a scientific study that would clearly track the response of the river to any one management practice or disturbance. Despite this, patterns and trends in the distribution of plant species were recorded.

The vegetation transects analysed to date indicated three main zones of vegetation along channel cross-sections: channel, riparian and hillslope. Establishment of these zones was clearer two years after the fire than a year earlier, and was most clear where alien species were rare. The channel and riparian vegetation appeared to be establishing itself more quickly than did the hillslope vegetation. Additionally, areas originally under pine recovered more rapidly than those under *Acacia*, probably at least partially because the *Acacia* re-grew far more vigorously than did the pines and because pines do not change the pH and nutrient status of the soils (Van den Berckt 2002). Even in the areas of heaviest *Acacia* infestation, however, indigenous species appeared along the channel margin, that is, in the poorly defined channel/riparian zone. The reason for the difference in recovery rate of the native flora along the channel margin and on the hillslope is not known.

Within the channel, the disturbances from fire and alien clearance were compounded by the winter floods. The 2001 winter was particularly wet (Reason *et al.* 2002) and floods inundated banks recently cleared of vegetation. Main Site 4 was particularly susceptible to these flows as it had high, almost vertical sandy banks. Edward Akunji, of the Department of Environmental and Geographical Science at UCT, completed an Honours project on this area (Akunji 2002), to answer our question: what was the origin of the large plug of sandy soil lying across the river at this point in the catchment? He identified the sand as mostly quartz and suggested the surrounding sandstone to be the parent material. The profile of the sandy bank was stratified with alternating layers of lighter and darker grey soil. The lighter bands had lower organic percentages than the darker bands. The sand in the plug is different to the surrounding reddish soil of the Peninsula Formation, delineating the deposit quite clearly from the surrounding slopes. Akunji discounted an Aeolian origin for the sand and concluded from the morphometry (structure) and lithology (physical character) of the grains that fluvial processes were responsible for transporting the sand to its present position. The bulk of this sandy material therefore derives from a whitish member of the Peninsula Formation situated upstream in the upper reaches of the Silvermine River. In the absence of carbon dating it is not possible to determine the age of the deposit. The stratified nature of the deposit led Akunji (2002) to conclude that development of the fan was not the result of a single depositional event, but rather resulting from a presently active and continuous process alternating between wetter and drier periods. Better understanding of the area is important as it is a source of considerable sediment input into the river, smothering downstream habitats and depositing sand all the way downstream to Clovelly Country Club and the artificial wetlands created by the City of Cape Town near the coast. This topic is re-visited in Section 14.6.

Despite the collapsing sandy banks, the composition of substrata at Site 4 remained much the same over the two summers of data collection. This may be because the site was already very unstable and mobile before the project began, with high amounts of sand already present. Evidence that the amount of sand present was abnormal was taken from King & Schael (2001), who tabulated the proportions of different sized substrata in sites on 18 relatively undisturbed Western Cape mountain stream or foothill zones. For all foothill sites but one, the proportion of sand was less than 2%, whereas in Main Site 4 on the Silvermine it was 47% in 2001 and 43% in 2002 (Table 14.12). Main Site 5, immediately downstream, still retained high levels, with 20% and 29% in the two years, whilst Main Site 2 upstream exhibited a high 33% in 2001 but a considerable reduction to 14% in 2002. The reduction in 2002 was likely due to an increase in vegetal cover in the surrounding and upstream catchment with time after the January 2000 fire, and the lack of this reduction at Sites 4 and 5 may well have been due to the presence of the sand plug.

The similarity of substrata and flow types in the two years of study is mirrored in the results of the invertebrate analyses. At Main Site 4, the invertebrate assemblages were to some extent separated by habitat, and to some extent by year, but the pattern was not precise. This may have been partly due to identifications being only to family level, but also probably partially because of the mobile and unstable nature of the channel at this point. The appearance of a distinct assemblage in fast water on stones and bedrock in 2002 is made up of four samples, all of which were collected upstream of the meander cut-off. These were the only four samples collected from fast flowing/rocky substratum during 2002.

Abstraction of water from the upper reaches at the weir (flow site 3) noticeably impacts flow in the river. All the flow is diverted into the weir at flow site 3 during the lower-flowing summer months, with surface flow only re-appearing below the first tributary (flow site 7). The signs are that the river was once perennial, as seepage does result in perennial surface flow downstream of flow site 7. The diversion of water will hamper natural rehabilitation of the aquatic and riparian assemblages in the next 2 km stretch of river, as many species common to these kinds of systems need surface water.

There are no known sources of pollution in all but the lowest part of the catchment, other than litter from Ou Kaapse Weg. The basic water-quality monitoring undertaken does not show major discrepancies from the guidelines for rivers in the south-western Cape in Dallas (*et al*. 1998). In the absence of historical records there are no other benchmarks against which the measured values can be compared.

14.6 Conclusion and recommendations

Different sections of the catchment are responding in different ways to the clearing methods. Recovery of indigenous vegetation appears to vary depending on the invading species being cleared. Recovery in old pine areas has been more successful than in areas under *Acacia*. This observation is supported by the findings of Cilliers (2002). Native riparian and aquatic vegetation appears to be recovering more rapidly than that on the hillslopes. This may be due to downstream dispersal of propagules from undisturbed, upstream communities, or simply due to growth being aided by the availability of water. It seems likely that recovery of riparian vegetation will occur successfully without planting, if the river corridor is protected from disturbance and if the *Acacia* and other alien species continue to be removed. Cleared plants should be removed from the riparian zone, that is, to outside of the macro-channel and thus out of reach of the largest floods. Stacked material in the riparian zone will retard rehabilitation of native species and can wash into the river during storms, possibly causing log-jams and becoming points of further bank erosion.

Erosion of the sand plug at Main Study Site 4 is likely to continue during periods of high flow. Since Main Study Site 2 has shown decreasing proportions of sandy substratum with time after the fire but Sites 4 and 5 have not, it seems probable that the sand plug is retarding rehabilitation of the middle and lower river. If Akunji (2002) is correct in his assumption that deposition at this point is a continuing process, then it seems possible that the plug built up over time, with an over-topping wetland, due to some natural hydraulic control downstream. The bands seen in the deposit could reflect climate-related times of greater or less deposit of sand or growth of plants. Loss of the natural wetland plant cover as *Acacia* infested the area could be leading to the river now eroding through the sandy plug. A similar situation has occurred in the Hout Bay River as reported by Pienaar & Boucher (1998), where vertical eroding banks of an ancient sandy deposit and wetland now threaten properties on both sides of the middle reaches of the river. It is recommended that further geological investigations be done, to ascertain if the sand plug is a natural feature of the Silvermine catchment, and if the reason for its original accumulation (e.g. a hydraulic control) is still in place and thus encouraging further deposition. This should be followed by a management assessment of whether or not removal of the plug and re-grading of the banks would be viable and of benefit to all the lower sections of the river. It might, for instance, be less costly to remove the plug than to embark on a continual dredging programme of the downstream artificial pools and wetlands.

A further area requiring management decisions surrounds the ancient dam and weir in the upper catchment. It could be argued that the dam has been in place for more than 100 years and the downstream channel has adapted to this. If removal of the dam was contemplated, impacts on the urban and recreational developments in the lower reaches would have to be assessed first. The weir has an obvious negative impact on the river by diverting all flow at that point in summer. Halting run-of-river diversions at this point, at least during the dry months, would enhance natural recovery of the downstream river.

15 EMERGING ECOLOGICAL AND GEOMORPHOLOGICAL PRINCIPLES FOR RIVER REHABILITATION

15.1 Introduction

There is general consensus that the science of river rehabilitation is still in a pre-paradigmatic state (Ractliffe & Reinecke 2002), with a weak conceptual basis to guide the plethora of procedures being undertaken. Books and scientific papers with titles referring to principles for river rehabilitation are becoming available, but most describe, some in a fairly unstructured way, guidelines, tips and general wisdoms about rehabilitation, leaving the more generic principles open to speculation. Where principles are given, the focus is mainly on a wider set of management principles, such as ensuring that objectives are realistic in terms of budget, or that stakeholders are involved. Such principles, which are often more like guidelines, are vital for a successful rehabilitation effort, but nested within them should be a sub-set of ecological and geomorphological principles that will guide the scientific aspects of the work. Without these, rehabilitation efforts stand the risk of being random experiments that make little or no contribution to scientific advancement or technical expertise.

In this Chapter, we list and discuss what appear to be some emerging scientific principles, whilst recognizing that many more may appear with time, and several could eventually be refined or become part of more encompassing principles. It is assumed that the wider management principles are also being addressed outside of these, and so these are not dealt with here, and that any of the scientific principles suggested here may well have to be modified to some extent due to the constraints of human activities within the catchment. These constraints, whatever they may be in any one situation and however they may require modification of the principles, are also not referred to again here. The main documents consulted were Kern (1992a); Newbury & Gaboury (1993b); Brooks & Sear (1996); Hobbs & Norton (1996); Stanford *et al.* (1996); Waal *et al.* (1998); and Hobbs (2002), with contributions from staff of this project.

15.1.1 What is a principle and why are they necessary?

According to the dictionary at www.dictionary.com, a principle in the context relevant here is:

- a basic truth, law or assumption;
- a rule or standard;
- a rule or law concerning the functioning of natural phenomena.

Principles may be seen as emerging from the general body of wisdom, through experiment, observation and hypothesis testing. Initially, "accepted generalisations or fundamental assumptions about the character or functioning of systems, or about the laws of cause and effect, form the basis of a chain of reasoning upon which hypotheses are formulated" (Ractliffe & Reinecke 2002). The testing and rejection or acceptance of hypotheses lead to the development of a set of intertwined concepts that comprise the accepted laws or principles in that field (Ractliffe & Reinecke 2002). Sound practice should be guided by such principles which, when combined with experience, may produce practical guidelines for managers. Such guidelines take time to evolve from the growing general wisdom in the field. Here we list four groups of suggested principles relevant to rehabilitation: general scientific, landscape, geomorphological and ecological. It is acknowledged that there could be many more. Added in italics beneath some entries are points of relevance with the three study rivers reported on in Part Three of this report.

15.2 General scientific principles of river rehabilitation

15.2.1 Rehabilitate back toward natural

Undisturbed systems are generally recognized as the most efficient users of scarce resources (Galat 2002). Further, where interventions are halted or removed, rivers will tend to revert to their original form and function, to the extent that this is now possible. Rehabilitation along this trajectory will be working with the river rather than against it, which should reduce maintenance efforts and costs as well as having ecological benefits for native species.
The strategy adopted by the Cape Peninsula National Park for the Silvermine River (Chapter 14) is to allow natural recovery to take place with no rehabilitation activities other than removal of alien vegetation.

15.2.2 Treat causes rather than symptoms

As with human health, treating the root causes of poor condition will bring more sustainable improvement than continual treatment of symptoms. For example, addressing soil erosion in the catchment may initially be costly but is more sustainable in the long term than continually dredging the river of sediments.
If the vertical sandy banks upstream of the Kuils River study site (Chapter 13) were stabilized to prevent bank collapse, continual dredging of the river channel could probably be reduced.

15.2.3 Rehabilitation is an interdisciplinary activity

Within the wider body of people involved, the scientific aspects require input from a number of specialists, such as hydrologists, geohydrologists, fluvial geomorphologists, sedimentologists, hydraulicians, water chemists, ecologists (particularly for fish, vegetation and invertebrates and maybe for water birds, herpetofauna and semi-aquatic mammals) and microbiologists.

15.2.4 All major abiotic drivers of ecosystem processes should be in harmony with the rehabilitation objectives

Hydrological, thermal, chemical and physical factors largely dictate the biological nature of an aquatic ecosystem, and desired results may be jeopardized by not addressing them all. For instance, recreated physical habitat may not support desired species if the water chemistry and temperature are inappropriate. Created physical habitat may also fail if appropriate flows or other fluvial processes are not available to maintain it, or if the hydraulic nature of the habitat does not suit the targetted species. Re-instatement or maintenance of primary abiotic processes should be addressed, and emphasized over mere structural modification.

15.2.5 Natural recovery may not achieve objectives

With degrading influences removed, natural recovery is a cheaper option, and has a good chance of success as long as a degradation threshold (e.g. species lost, communities drastically changed) has not been crossed. If such a threshold has been crossed, natural recovery may well be slow, partial or not occur and intervention could be needed. For instance, if native species that cannot move between catchments have been lost from a system, re-introduction would be necessary.

The reaches downstream of the sand plug in the middle reaches of the Silvermine River may represent a case of slow partial recovery, where active intervention to remove the sand plug itself might be a feasible management option.

15.2.6 It is easier to cross a degradation threshold than to return over it

Once a degradation threshold has been crossed, it may require massive management inputs to return it toward a more natural condition. The thresholds are seen by some as linked to the functional composition of the ecosystem (e.g. wetlands in the landscape that feed a river system), and where this has changed (e.g. wetlands eradicated and changed to agricultural land), transition back is usually very difficult. Two challenges for restoration ecology are to understand the variables that need to be manipulated to achieve these transitions, and to develop practical and cost-effective methods for forcing the transitions. For instance, if lake aquatic vegetation used to switch between two functional forms (e.g. macrophytes and phytoplankton) but now is permanently in one state (phytoplankton), major management inputs in terms of (say) reducing nutrients may be needed for the system to revert to its original two-state form.

Main Study Site 4 on the Silvermine provides an example of one or maybe even two degradation thresholds that have been crossed (Section 14.6). The first is the loss of native vegetation by the severe infestation of aliens. The second is the possible consequent erosion of the riverbed down through an ancient wetland (this is speculation at this stage). It would be far more difficult to pass back through the second threshold than the first because of the need to reconstitute the wetland and fill in the entrenched channel, but both would require active management intervention.

In the case of the Lourens River, many of the problems that are now becoming apparent in its lower reaches have arisen from the gradual transition of the areas of natural vegetation within the 20-year flood-line into urban landscapes. This has resulted in catchment hardening, increased flood peaks in and degradation of the lower reaches, and urban encroachment into what may have been a wider meandering, swampy lower river where overbank flooding was a natural phenomenon. To revert this area toward natural is almost impossible because of its urban context, and would be extremely costly. Management of the present degraded situation is probably also costly.

15.2.7 Ecosystems are dynamic and can naturally exist in alternative metastable states

Ecosystems are dynamic, naturally changing over all time scales. Major natural fluxes may result in transitions between alternative metastable states. Such transitions, which perhaps pass through major thresholds (e.g. a river that is flowing or dry), can be caused by management actions but also may occur naturally due to environmental factors, with the occurrence of different states depending on particular combinations of the driving forces. The natural dynamism and the alternative states have implications for rehabilitation goals, with ‘naturalness’ having to be carefully defined and understood in the context of the forces that will aid recovery at the site.

15.2.8 Monitoring is an essential component of rehabilitation

In any scientific work, monitoring the outcome is essential. In rehabilitation work it is particularly important because of the early state of understanding in restoration ecology. Without monitoring there may no documentation evaluating a rehabilitation project. Rehabilitation science can only grow with such shared knowledge.

Monitoring of the Silvermine River over the last two years has already produced new insights into the recovery of native riparian vegetation after fire and alien removal. Monitoring of the Kuils River over the same time span has shown upstream degradation due to bank collapse that correlated with a programme of downstream channel widening.

15.2.9 Rehabilitation operates over a range of spatial and temporal scales

Some parts of the ecosystem will respond quickly to rehabilitation activities, others more slowly. In general, channel responses could take decades whilst short-lived plants and animals could respond within weeks. Predicted outcomes and evaluation should be geared to both short and long-term time scales, so that goals are clearly linked to these and interest does not flag.

15.3 Principles at the landscape level

15.3.1 Catchment-level processes affect local form and function

Sites targeted for rehabilitation cannot be understood without reference to their broader spatial context, obtained through catchment analysis, geomorphological and ecological analysis, identification of the processes leading to degradation, and with reference to the broader temporal picture. Morphology of the channel, for instance, is directly influenced by the nature of the sediment load from the surrounding catchment and so river rehabilitation schemes need to address long-term trends in sediment delivery and an appreciation of how this affects channel morphology. The rehabilitation team should know where the sediment comes from and address any problems at their root.

15.3.2 Assessment and planning should start at the catchment scale

Few explicit guidelines exist for landscape-scale assessments, as most documentation is on site-specific rehabilitation. Some issues that have to be addressed at this broader level, apart from those already mentioned in previous sections, include: linkages in the landscape, including connectivity for migratory species; retention of limiting resources; protection, maintenance or creation of spatial heterogeneity; protection of existing habitat patches as, *inter alia*, propagule donor patches; and wind and animal dispersal of desired seed. Sites should not be rehabilitated if other parts of the river system cannot support the desired outcome.

15.3.3 Rehabilitation should progress from source to mouth, and include tributaries

Rehabilitation of isolated sites can fail due to the influence of upstream, degraded areas. Rivers are longitudinal ecosystems, and upstream reaches directly influence the nature of all downstream reaches by delivering, for instance, pollutants, nutrients, sediment and organisms. Upstream reaches may be degraded in terms of altered sediment and hydraulic regimes, and if rehabilitation starts at the mouth and works upstream, there could ultimately be destabilization of the whole river system including reaches presently in good condition.

15.4 Geomorphological principles

15.4.1 Channel features are a product of their catchments – work with the river not against it

Many schemes are driven by perceptions of 'natural' rivers taken from other geographical areas, which do not consider conditions within the catchment of concern. This leads to the creation of inappropriate channels and habitats relative to the geomorphological processes that occur in a system. The inherent unsustainability of these recreated features (e.g. riffles in lowland sandy rivers, which soon become silted) compromises both the success of the project and the chance of future rehabilitation endeavours being approved.

Channel geometry at any point along a river system is a reflection of the catchment variables influencing that point. Designed channel shape, size and features should be based on adequate analysis of the geomorphological characteristics of the catchment and river, such as the underlying geology, and the patterns of water and sediment transport that create and maintain the channel and floodplain morphologies, so that rehabilitation works with the natural character of the river rather than against it.

In the same way, banks should be graded according to their material, with steeper banks in more cohesive material and flatter banks in sandy material.
Knowledge of the angle of repose in the different bank materials is essential for re-structuring of banks. Eroded land upstream of the widened reach in the Kuils study site was replaced mechanically after the winter floods. The restructured banks were extremely steep, which may lead to their further collapse with the next major flood.

Reference reaches, as long as they are relatively natural, can be used to provide relevant channel morphometry and guide restoration measures that are in keeping with the catchment.

15.4.2 Different river zones have different abilities to mobilize sediments – let the river do the work

Steeper reaches of rivers have more stream power than flatter reaches, and so can mobilize larger volumes and particles of bed material. Substrata in the rehabilitation reach should be sized according to undisturbed reaches in similar rivers and zones of the same catchment with similar flow regimes. The degree of intervention required to restore the channel decreases where there is appropriate stream power to mobilize sediments and an appropriate supply of sediments.

15.4.3 Dynamic channels are more natural than restricted ones

Channels naturally change in shape, position and features over time – some more than others. Channels that remain dynamic in these respects may migrate laterally, causing natural erosion but also creating a diversity of habitats. Lateral erosion may come to an end in a natural way when the river reverts back to its original position by shortening its loops or meanders. Channels allowed to maintain a dynamic equilibrium may need more space than those confined by human intervention but often need less maintenance and are more efficiently functioning ecosystems.

It is crucial to restrict development to the area outside that in which a river meanders naturally. The Lourens River, during the winter season of 2001, migrated laterally at Site One into a developed agricultural area just downstream of an artificial gabion that confined its channel. Constriction of its dynamic nature at one place simply moved the lateral migration to another area. In this way an erosion problem may have been solved at one place, but it was transferred to another. Concreting the channel to stabilise its position is only a solution if society is willing to accept the complete loss of the river ecosystem at that point and its degradation to an unknown degree at all points downstream.

15.4.4 Straightening and smoothing channels increases erosive power

Channels that are straightened and smoothed have increased gradients and current speeds. This may cause erosion, in reaches both upstream and downstream of the modifications. Widening can likewise cause upstream erosion, due to a local increase in the friction gradient. Where sinuosity is re-introduced, this should be based on channel slope, stream power and sediment load. If any of these are out of balance, the river will respond by either incising its channel vertically or laterally or by depositing suspended sediments.

After channelisation and regrading in 2000 of Study Site Three in the Kuils River, erosion occurred upstream with a considerable loss of land and steep unstable banks. When Site One was later regraded and widened after the floods in 2001, the same thing happened upstream. There are signs that more erosion in this area over the winter season of 2002 may lead to undercutting of the old Bottelary Road bridge (Chapter 13)

15.4.5 A range of flows is required to maintain a channel

Different sized flows perform different functions in channel maintenance: almost all flows except the smallest 'work' the channel material in some way. Floods with long return periods maintain the macro-channel, whilst those with around a 1.5 to 5-year return period (in different kinds of systems) are sometimes termed the bankfull discharge and recognized as maintaining the active channel. Small intra-annual floods may be most important for sorting bed substrata, and so be important agents for maintaining habitat heterogeneity (King & Schael 2001), whilst a range of higher flows flush out fines from in-channel pools. An appropriate mixture of flows is thus needed for maintenance of the channel and habitats.

15.4.6 Floodplains, wetlands and groundwater are part of the river ecosystem and should be included in rehabilitation projects

Floodplains, wetlands, groundwater and the riparian zones are all components of the river ecosystem, and contribute to varying degrees to its pre-impact character. Rehabilitation of the channel alone, without due attention to these wider ecosystem components, may not produce the desired result due to the loss of these other functional parts of the ecosystem. Habitats in all these areas boost biodiversity by increasing physical and hydraulic heterogeneity, with subsequent benefits in terms of enhanced plant and animal communities. Where loss of these ecosystem components or their reduction below viable size could jeopardise rehabilitation efforts, consideration should be given to their re-creation.

15.4.7 Sites with all their potential geomorphic units offer the optimal diversity of aquatic habitat

Many modified channels, and even rehabilitated ones, have simple trapezoidal cross-sections. Re-created channels with multiple stages (levels) allow low flows with some depth to move along a small active channel thus, for instance, creating conditions for fish survival and passage. They also present much better conditions for bank stability. Higher flows spill onto terraces, allowing small floodplains to develop. Substrata are distributed according to the mosaic of hydraulic conditions. Heterogeneity of habitat is enhanced as is, potentially, biodiversity, and organisms become distributed according to their needs in terms of hydraulic habitat, substrata and inundation.

15.5 Ecological principle*s*

15.5.1 Biotic as well as abiotic factors determine the communities that a river can support

With all abiotic variables in harmony with the rehabilitation objective (Section 15.2.4), biotic factors may still confound it. Species have different habitat and life-cycle requirements, which dictate their distribution at every level from continental to microhabitat. Sites in the mountain-stream zone of a river in the fynbos biome, for instance, will not support communities from a Natal river, or even from the foothill zone of the same river, although individual species may be able to survive. Similarly, plant communities that normally occur high on the macro-channel bank and that are inundated very infrequently will not thrive if planted in the active channel. Sites that are to be stocked should thus be assessed in terms of their biogeographical affiliations, their biological zone along a river and the shape of their cross-sectional profile, and appropriate species introduced to appropriate parts of the site. Native species of that region, and if possible that catchment, should be used unless there are specific objectives that invalidate this, as introduced species have the potential to radically impact existing indigenous populations. Where the likely impact of an alien species is not known, the species should not be introduced: the precautionary principle should apply.

15.5.2 Maximizing habitat for one species is not the same as re-creating the biotic structure and functioning of a stream ecosystem

Different species, even within the same community, have different life requirements. Some fish species, for instance, may only spawn during drought years, whilst other fish species within the same community may fail to spawn in drought years. Their continued survival within the same community depends on varying environmental conditions that favour some species one year, and others in another. Attempts to continually supply the optimal conditions for a target species will ultimately be to the detriment of the wider community of which the target species is an integral part. It is a better strategy to mimic, to the extent possible, all aspects of the natural abiotic heterogeneity of the site.

15.5.3 Re-colonisation of damaged areas will occur at levels dependent on the habitats available, stocking sources and remaining stressors

Rehabilitated areas will be re-colonised, providing conditions are not adverse for all species. For instance, most colonisation by aquatic invertebrates will probably occur from downstream drifting organisms, though aerial dispersal and active swimming could be other important routes. Creating known favoured habitats enhances the chances of re-colonisation by desirable species, though the limitations of which habitats could be sustained at that point (Section 15.4.1) must be addressed. Desired species that have been eradicated from the system, with no means of migrating back into it, would need to be re-introduced.

15.5.4 The rate of recovery is dependent on the ecosystem and the intensity of the disturbance

Aquatic habitats and communities may recover from disturbance more quickly than terrestrial ones, providing the basic structure of the environment has not been destroyed. Riparian and marginal communities may recover more quickly because of their easy access to water, for example, whereas the communities higher up the bank may be slower to respond due to their drier conditions. Conversely, high-altitude wetlands where the basic structure and drainage pattern of the peaty soils has been destroyed (e.g. by vehicle tyres) may take decades or longer to recover even though water is still freely available.

Areas lightly infested with aliens have a better chance of speedy recovery than heavily infested areas, and rehabilitation efforts would probably profit by first focusing on the former.

15.6 Conclusions

This is a preliminary list of suggested principles, which could grow with time to become a set of principles through input from others and modification. Above this are other, more fundamental, principles or concepts of ecology, which bear on the science of rehabilitation. These would address such aspects as what constitutes a community; what constitutes the functioning of a river ecosystem; whether replacing physical habitat can restore functioning; and whether alien species can play the same role in functioning as lost native ones. These issues have not been attempted here.

16 CONCLUSIONS AND RECOMMENDATIONS

16.1 Conclusions

River rehabilitation is characterised by a dearth of testable hypotheses, and consequently an absence of rehabilitation principles (Ractliffe & Reinecke 2002). As a new science, its development depends on accumulating empirical observations, using these to develop and test hypotheses, and from that tentatively beginning the development of principles. In this WRC project, it was possible to follow part of that sequence, by empirical observation of how three rivers responded to management interventions. A review of the international literature allowed further understanding, by linking some of these observations to wider rehabilitation knowledge. The principles presented in Chapter 15 could not emerge from this via hypothesis testing because it was not possible to set up such a research programme within the financial, time and management constraints faced. Thus, the hypothesis-testing phase was missed, and the principles were instead drawn from an understanding of river ecosystems gained before and during the project and from scattered information in the international literature. This is a first attempt to present a set of principles as some relevant over-arching wisdoms, and it became clear that the list could be almost any length. Time and testing will prove in an iterative manner which of the principles are valid, which are not particularly relevant or true, and what is missing.

Ractliffe & Reinecke (2002) point out that there are a multitude of rehabilitation projects being undertaken worldwide, each of which could be viewed as an experiment from which considerable knowledge could be obtained. Each such experiment could be: designed to test one or more hypotheses; objectively documented; evaluated; and the results of the evaluation widely disseminated. This would greatly aid the structured development of a set of rehabilitation principles.

To support this process, a co-ordinated programme of gathering ecological and geomorphological data on rehabilitation is needed that will provide the theoretical basis for formulation and testing of hypotheses. This is particularly important because river response to intervention is slow and so understanding will grow slowly anyway. Such a programme would be most effective if run over a number of countries with a central co-ordination centre, and in its simplest form could perhaps merely provide a means of guiding rehabilitation exercises in a way that allows hypothesis testing to take place. Monitoring could then be used to assess the validity of the hypotheses, with the results fed through to the central point responsible for principles development. Managers would benefit though a structured growth of understanding and guidelines for rehabilitation.

16.2 Recommendations

16.2.1 Create a centre of scientific expertise on river rehabilitation within South Africa

Research and learning on river rehabilitation would best develop in an environment where the few scientists involved in the field could collaborate, plan joint ventures and exchange knowledge. Such a group is already forming, supported by the Water Research Commission. This could become South Africa's contribution to a wider international Centre of Expertise.

16.2.2 Link up with similar groups globally to develop hypothesis-testing rehabilitation projects

A number of key projects, worldwide, could be used to test hypotheses and to develop the scientific principles of rehabilitation. Such projects should address large-scale rehabilitation under controlled situations, where multiple management practices cannot obscure river response.

16.2.3 Provide structured funding for rehabilitation of rivers and evaluating the outcome

Structured documentation of the success or failure of rehabilitation projects is the major and most vital way to develop understanding. Funds for monitoring and evaluation should be an essential part of any rehabilitation activity, and should include supporting development of the whole poorly understood and developed science of monitoring.

16.2.4 Develop a scientific research programme on river rehabilitation

In addition to monitoring rehabilitation projects, fundamental research should begin on a range of topics. The list is potentially very long, and the national scientific centre of rehabilitation expertise should identify priorities in liaison with managers.

16.2.5 Synthesise findings of the research and monitoring activities and further develop a set of rehabilitation principles

This would be an iterative process.

16.2.6 Develop guidelines from the principles

With growing evolution of the principles and experience in the field, practical guidelines on river rehabilitation that reflect accepted scientific wisdom and local needs should be written for managers.

16.2.7 Develop close liaison between scientists and managers

The principles and guidelines should be developed in close liaison with managers. At present there is extremely limited contact between South African river scientists and the authorities that normally manage the rehabilitation or 'improvement' of rivers. Closer liaison could help encourage rehabilitation projects with soft engineering solutions and, to the extent possible within budget and other constraints, which allow hypothesis testing.

16.2.7 Develop appropriate policy and regulatory frameworks

These (and successive) principles and any future guidelines acquire meaning only if supported by appropriate regulatory frameworks and policy. They can additionally serve to identify areas that need to be addressed in existing policy and institutional arrangements relating to river rehabilitation (e.g. National Water Act Chapter 4). Relevant authorities and institutions should be encouraged to take note of the emerging principles, to contribute toward their further development, and to use them to inform and where necessary modify current and future policy and management decisions.

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APPENDIX: 12.1 **Percentage cover of plant species at Site 1, Transect 1.3 in 2001.** Data codes: 15-01: = quadrant #, 01 = year.

Vegetaion data codes	01-01	02-01	03-01	04-01	05-01	06-01	07-01	08-01	09-01	10-01	11-01	12-01	13-01	14-01	15-01	16-01	17-01	18-01	19-01	20-01	23-01	24-01	25-01	26-01	27-01	28-01	29-01	34-01	35-01	36-01	37-01	38-01	39-01	40-01	41-01	42-01	43-01	44-01	45-01
Pennisetum clandestinum	40	20	35	5	5	6	5	15	20	6	10	0	5	15	20	40	53	0	10	10	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tropaeolum majus	0	0	0	5	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Populus x canescens	0	0	0	0	20	20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Acacia mearnsii	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scirpus species	0	0	0	0	0	0	0	0	6	20	0	0	0	0	0	0	10	30	20	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago lanceolata	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	2	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Conyza canadensis	0	0	0	0	0	0	0	0	0	0	4	3	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Distephanus angolensis	0	0	0	0	0	0	0	0	0	0	10	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dittrichia graveolens	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Convolvulus arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	15	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0	40	60	75	0	30	10
Commelina benghalensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	0	0	3	0	3	10	20	2	10	50	90	99	99	60	40	20	0	0	0	0
Tarchonanthus camphoratus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachiaria eruciformis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	35	35	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cynodon dactylon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eustachys paspaloides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	10	0	6	20	40	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus longus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15	20	32	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polygonum species	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	10	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ipomoea purpurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	40	0	0	0	20	0	0

APPENDIX 12.2 **Percentage cover of plant species at Site 1, Transect 1.3 in 2002.** Data codes 15-02: 15 = quadrant #, 02 = year.

Vegetation data codes	01-02	02-02	03-02	04-02	05-02	06-02	07-02	08-02	09-02	10-02	11-02	12-02	13-02	14-02	15-02	16-02	17-02	18-02	19-02	20-02	21-02	24-02	25-02	26-02	27-02	28-02	29-02	34-02	35-02	36-02	37-02	38-02	39-02	40-02	41-02	42-02	43-02	44-02	45-02	46-02	47-02	48-02
Pennisetum clandestinum	80	90	90	60	0	0	3	15	3	5	25	5	5	5	5	25	60	70	75	15	0	0	0	0	0	0	0	0	0	0	10	30	20	30	5	0	0	10	5	0	0	10
Populus x canescens	0	0	72	75	70	70	70	70	70	70	70	70	90	90	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tamarix chinensis	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tropaeolum majus	0	0	0	0	15	20	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fumaria officinalis	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Conyza bonariensis	0	0	0	0	0	0	0	0	0	2	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hypochaeris radicata	0	0	0	0	0	0	0	0	0	5	2	5	6	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago lanceolata	0	0	0	0	0	0	0	0	0	3	3	23	4	2	3	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lolium perenne	0	0	0	0	0	0	0	0	0	0	5	15	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oxalis pes-caprae	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rumex acetosella	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lactuca serriola	0	0	0	0	0	0	0	0	0	0	0	0	3	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Verbena officinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brachiaria serrata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus esculentus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	10	0	8	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus longus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	5	0	15	8	0	0	0	0	0	0	0	0	0	0	0	0
Commelina benghalensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	4	0	2	5	3	0	20	10	5	5	0	0	0	5	0	0	0
Polygonum species	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	3	3	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Polypogon species	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Panicum gilvum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rumex species	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	5	2	0	0	0	0	0	0	0	0	0	0	0	0
Convolvulus arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	50	90	0	80	0	90	0	80	0
Ipomoea purpurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30	60	0	0	90	0	60	0	90	0	60
Bidens pilosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0

Appendix 13.1 Macro-invertebrate abundance ratings for Sites 1 to 4. (Table 13.4 for Hydraulic biotope number and 13.5 for ratings.) Where S1-1-01 refers to Site (S1), sample number (1) and year (01).

Sample code	S1-1-01	S1-2-01	S1-3-01	S1-4-01	S1-5-01	S2-1-01	S2-2-01	S2-3-01	S2-5a-01	S2-5b-01	S2-6-01	S2-7-01	S2-8-01	S2-9-01	S2-10-01	S3-1-01	S3-2-01	S3-S3a-01	S3-3b-01	S3-4-01	S3-5-01	S3-6-01	S3-7-01	S3-8-01	S3-9-01	S4-1-01	S4-2-01	S4-3-01	S4-4-01	S4-5-01
Habitat	6/SM	20/RS	10/RS	8/BF	12/BF	1/SM	8/BF	8/SM	1/BF	1/BF	21/SM	11/NF	1/SM	8/BF	14/SM	11/BF	9/BF	11/NF	1/SM	2/SM	18/SM	15/SM	1/SM	14/SM	9/SM	24/SM	27/BF	26/SM	23/SM	28/SM
Sample size (m)	0.8x0.6	1.4x0.4	3x0.4	1.6x0.3	3x1	1.5x0.4	2.5x1.4	2.3x0.3	Core	Core	2.7x1	2.3x1	Core	1.6x0.7	1.5x0.5	1x0.5	1.2x0.5	2x1	Core	0.5x0.5	0.5x0.2	1.5x0.5	Core	1.5x0.5	2.7x0.5	1.5x0.5	1.5x0.6	1.3x0.4	2x0.2	
Hirudinae	2	0	1	0	0	2	0	2	0	0	0	2	2	0	0	0	0	0	4	4	1	1	1	0	1	0	0	3	0	0
Naididae	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	2	1	0	0	4	0	0	0	1	0	0	0	0	0	0
Coenogrionidae	0	0	0	3	2	1	1	4	0	0	1	2	0	2	0	0	2	0	0	0	0	0	0	0	2	0	0	0	0	0
Gomphidae	0	0	0	4	2	0	3	2	0	0	0	2	0	1	0	2	4	0	0	0	0	0	0	0	2	2	3	1	1	0
Aeshnidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Libellulidae	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Baetidae	0	0	0	2	2	0	0	0	0	0	0	2	0	1	0	1	5	2	0	0	2	0	0	1	2	0	1	0	2	0
Corixidae	0	0	0	0	0	0	0	0	0	0	2	4	0	2	0	3	0	4	0	4	1	2	0	0	2	0	0	0	0	0
Veliidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Gerridae	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	3	0	0	0	0	0	0	1	0	0	0	0	0	0
Gyrinidae	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	5	4	3	5	5	3
Chironomidae	0	4	4	0	4	4	4	3	0	0	3	4	0	4	4	0	2	0	0	2	3	0	0	0	5	5	3	5	5	3
Culicidae	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Lymnaeidae	0	0	0	1	0	0	0	3	0	1	0	2	0	0	2	2	0	0	0	2	2	0	0	2	2	0	0	0	0	0
Physidae	0	0	0	2	3	3	0	2	3	2	3	2	0	3	4	3	3	3	2	2	1	3	0	2	3	2	2	0	0	0

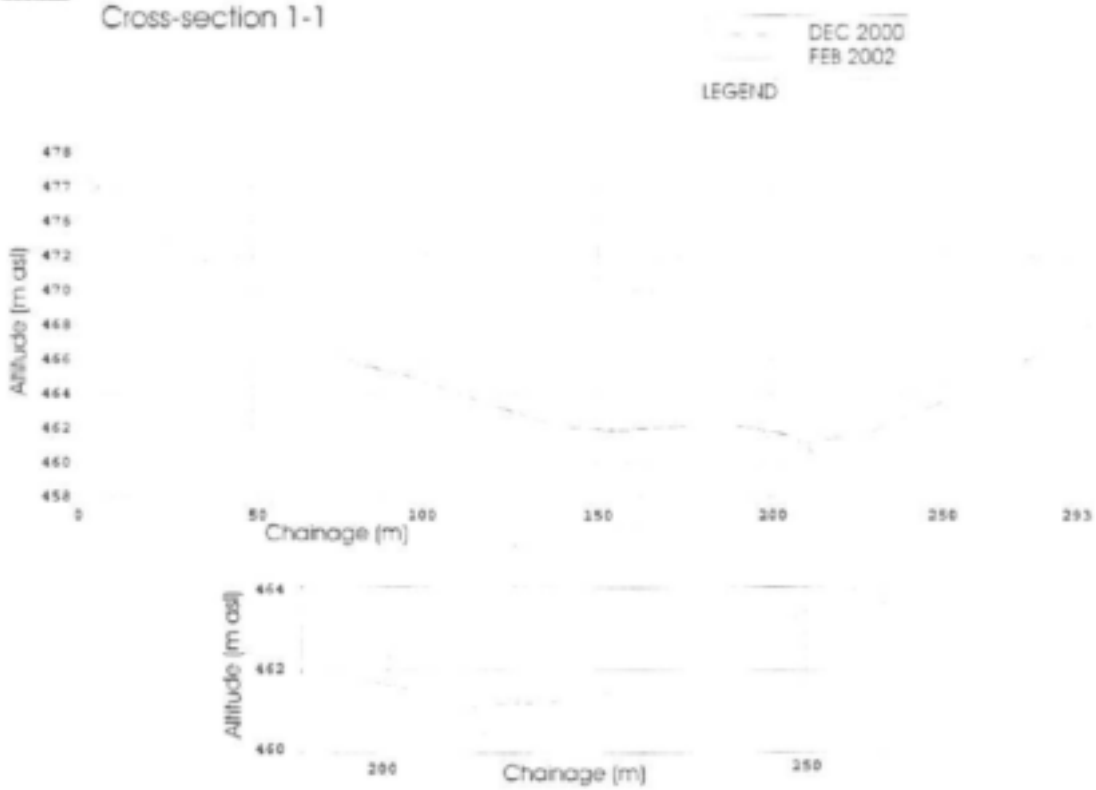
Sample code	S1-1-02	S1-2-02	S2-1-02	S2-2-02	S2-3-02	S2-4-02	S2-5-02	S2-6-02	S2-7-02	S3-1-02	S3-2-02	S3-3-02	S3-4-02	S3-5-02	S3-6-02	S4-1-02	S4-2-02	S4-3-02	S4-4-02	S4-5-02
Habitat	9/BF	4/SM	3/SM	4/SM	5/SM	1/SM	9/BF	3/SM	3/BF	9/BF	17/SM	2/BF	7/BF	19/RS	26/SM	26/BF	27/SM	25/SM	28/SM	25/SM
Sample size (m)	1X0.5	Core	0.5x0.5	2x0.5	0.5x0.5	Core	1x0.5	Core	Core	1.5x0.5	1.5x0.3	Core	2.5x0.6	1.5x0.4	1x0.2	1.4x0.4	2x0.2	1x0.2		0.8x0.2
Hirudinae	0	0	5	0	5	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Naididae	2	3	4	4	3	4	0	3	4	2	2	0	2	2	2	2	4	3	3	2
Coenogrionidae	2	0	2	0	1	0	3	0	0	2	0	0	0	0	1	0	2	0	0	0
Gomphidae	0	0	2	0	1	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0
Pleidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Naucoridae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Baetidae	0	0	0	0	0	0	3	0	0	3	0	0	4	0	0	0	3	0	0	2
Corixidae	0	0	0	0	0	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0
Veliidae	1	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Gerridae	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Dytiscidae	3	0	2	0	1	0	0	0	0	3	2	0	3	0	0	3	0	0	0	0
Simuliidae	0	0	0	2	0	0	0	0	0	0	2	0	0	2	2	0	2	0	2	0
Chironomidae	0	2	5	4	3	0	3	0	0	4	4	1	3	5	4	4	5	4	4	4
Athericidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Lymnaeidae	0	0	0	0	0	0	2	0	0	3	0	0	0	0	0	0	2	0	0	0
Physidae	5	1	4	3	3	0	3	2	2	2	2	3	4	4	3	4	3	2	1	4

Appendix 13.2 Percentage cover of plant species transect 1.2 for 2001 and 2002. Data codes 01.1.2.1 = year, site, cross section and quadrant numbers**.** All species are herbs except for *Acacia saligna* that is a tree.

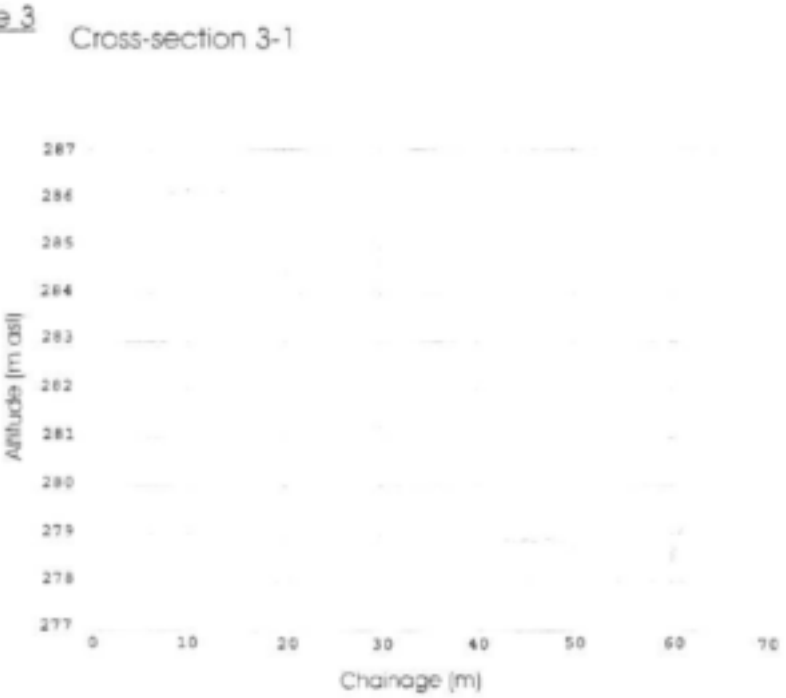
Transect 1.2-2001	01.1.2.1	01.1.2.2	01.1.2.3	01.1.2.4	01.1.2.5	01.1.2.6	01.1.2.7	01.1.2.8	01.1.2.9	01.1.2.10	01.1.2.11	01.1.2.12	01.1.2.13	01.1.2.14	01.1.2.15	01.1.2.16	01.1.2.17	01.1.2.18	01.1.2.19	01.1.2.20	01.1.2.21	01.1.2.22	01.1.2.23	01.1.2.24
Pennisetum clandestinum	5	5	2	0	2	20	80	99	41	0	0	0	40	50	70	50	99	50	15	10	5	20	20	10
Bromus diandrus	0	10	15	5	40	10	2	0	0	0	0	0	0	0	0	0	0	0	0	30	30	40	10	30
Chenopodium album	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus species	0	0	0	0	0	0	0	0	2	30	20	5	0	0	0	0	0	0	0	0	0	0	0	0
Polygonum aviculare	0	0	0	0	0	0	0	0	20	10	10	0	0	0	0	0	0	0	0	0	0	0	0	0
Isolepis fluitans	0	0	0	0	0	0	0	0	0	30	55	55	0	0	0	0	0	0	0	0	0	0	0	0
Xanthium strumarium	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus longus	0	0	0	0	0	0	0	0	0	0	0	40	10	50	30	50	0	0	0	0	0	0	0	0

Transect 1.2-2002	02.1.2.1	02.1.2.2	02.1.2.3	02.1.2.4	02.1.2.5	02.1.2.6	02.1.2.7	02.1.2.8	02.1.2.9	02.1.2.10	02.1.2.11	02.1.2.12	02.1.2.13	02.1.2.14	02.1.2.15	02.1.2.16	02.1.2.17	02.1.2.18	02.1.2.19	02.1.2.20	02.1.2.21	02.1.2.22
Amaranthus deflexus (sl)	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Echium plantagineum (sl)	15	8	10	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Raphanus raphanistrum (sl)	3	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tribulus terrestris	10	2	0	0	0	0	10	50	20	0	0	0	0	0	0	0	0	0	0	0	0	0
Amaranthus deflexus	0	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cynodon dactylon	0	10	0	0	0	0	5	15	5	0	0	0	0	0	0	0	0	0	0	0	0	0
Acacia saligna (sl)	0	0	3	0	5	8	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pennisetum clandestinum	0	0	5	5	10	0	0	3	8	2	30	85	15	0	0	0	0	10	30	88	75	72
Paspalum distichum	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyperus rotundus	0	0	0	0	0	0	5	10	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Datura stramonium	0	0	0	0	0	0	30	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Citrullus lanatus	0	0	0	0	0	0	0	8	52	95	55	0	0	0	0	0	0	0	0	0	0	0
Cyperus species	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0

Site 1

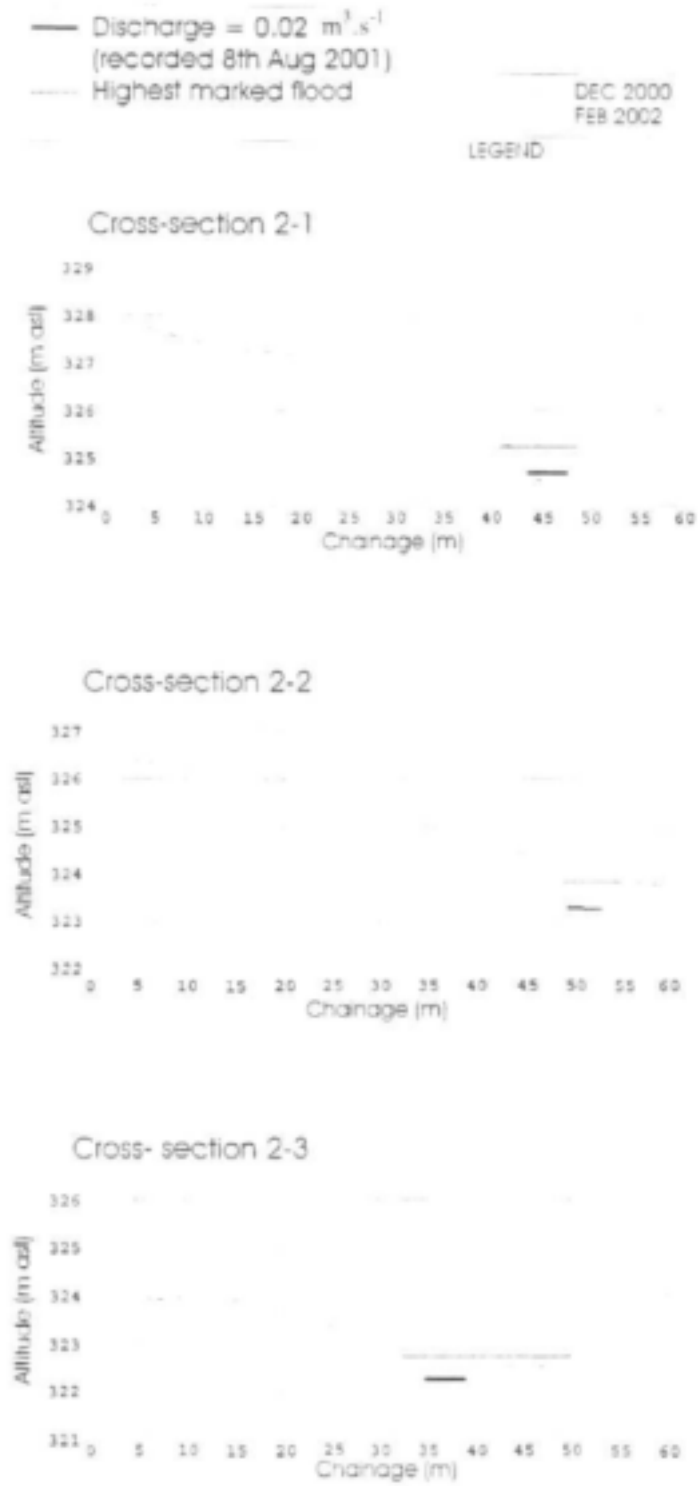


Site 3

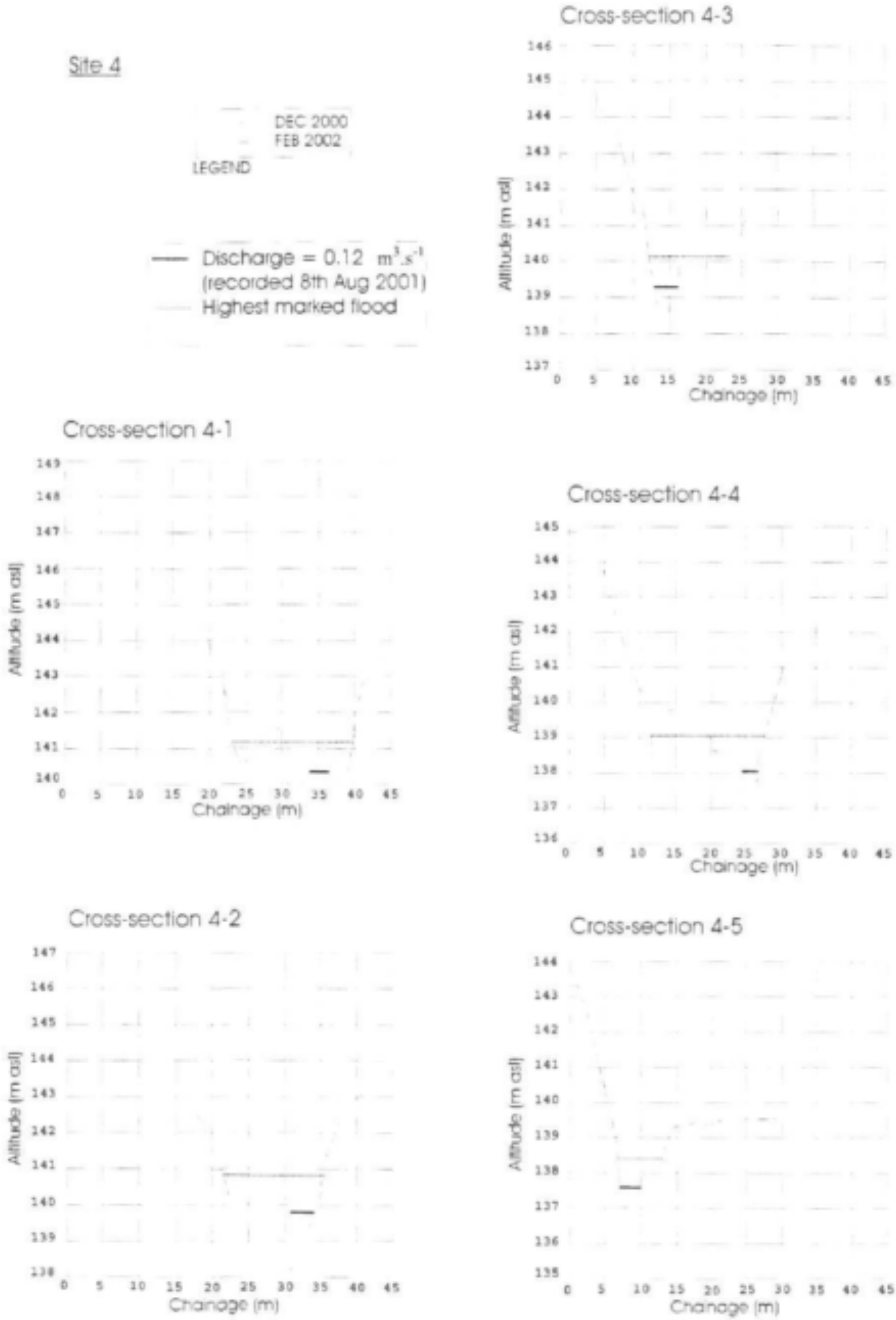


Appendix 14.1a **Surveyed cross-sections for Supplementary Study Sites 1 and 3.** The study sites can be located on Figure 14.2.

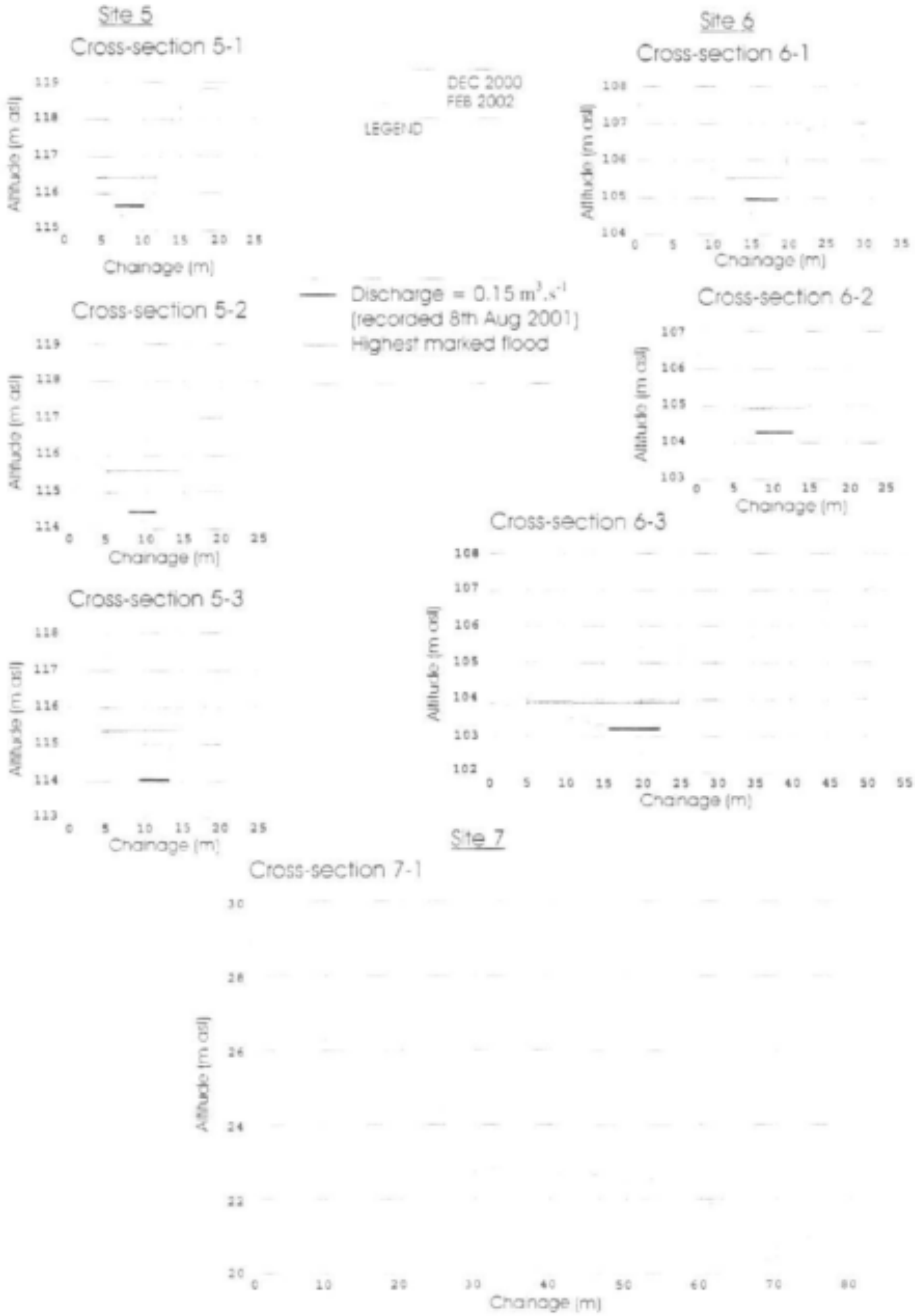
Site 2



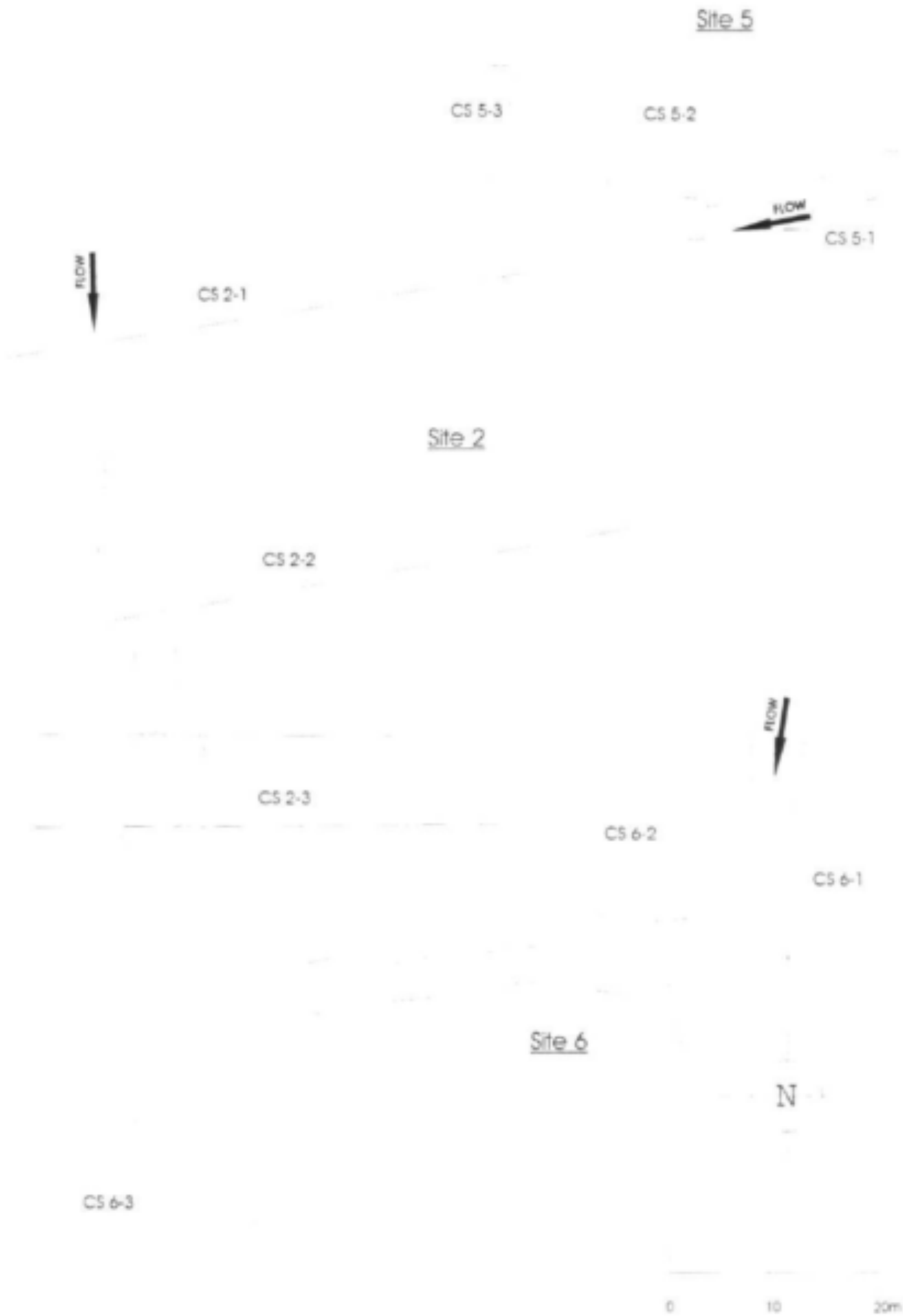
Appendix 14.1b **Surveyed cross-sections for Main Study Site 2.** The study site can be located on Figure 14.2.



Appendix 14.1c **Surveyed cross-sections for Main Study Site 4.** The study site can be located pm Figure 14.2.



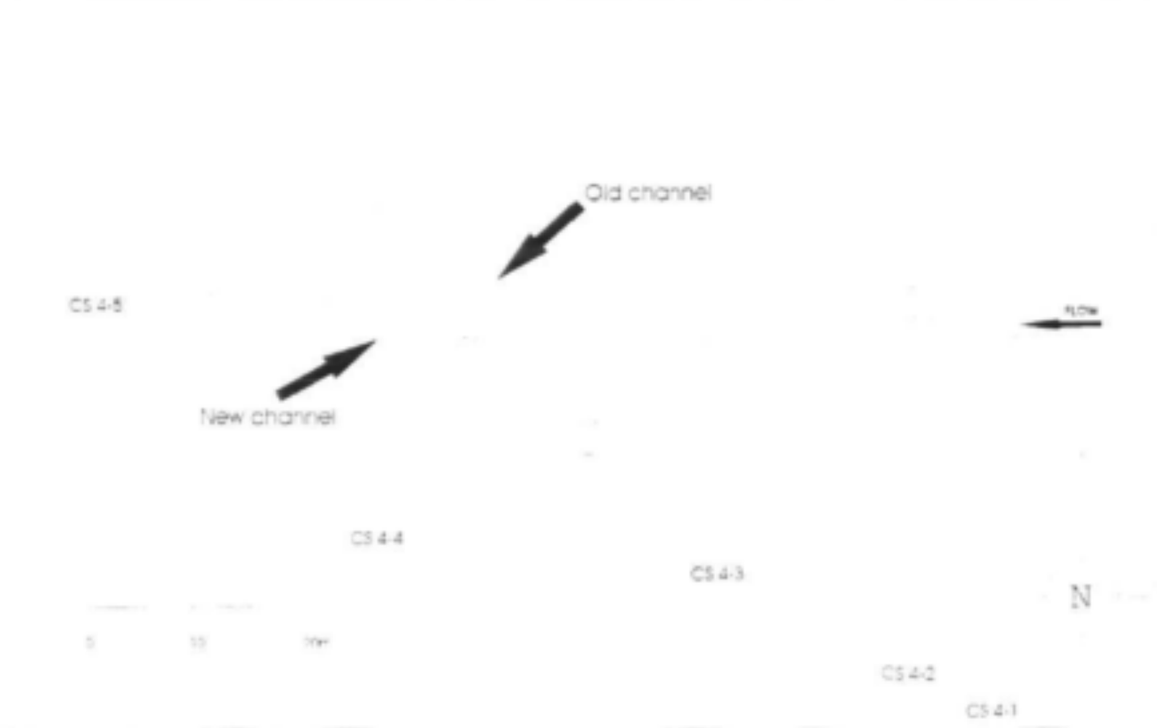
Appendix 14.1d **Surveyed cross-sections for Main Study Sites 5 and 6 and Supplementary Study Site 7.** The study sites can be located on Figure 14.2.



Appendix 14.2a **Channel outlines of Main Study Site 2, 5 (mapped December 2000) and 6 (mapped January 2002) showing location of surveyed cross-sections.** At least 70 points were surveyed for each of these maps.

a Channel outline December 2000 showing single wet edge at cross-section 4-4

b Channel outline January 2002 showing newly formed channel after the meander cut-off had taken place



Appendix 14.2b **Channel outlines of Main Study Site 4 showing location of surveyed cross-sections.** Map a (153 surveyed points) and Map b (220 surveyed points). This section of the catchment is the most unstable. During the winter high flows of 2001 a meander cut-off occurred at cross-section 4-4 (CS4-4).

Appendix 14.3a **The height and number of each surveyed point for cross-sections 1-1, 2-1 and 2-2, and the distances between surveyed Points.**

Cross section 1-1	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Rbank control	467.87	0.00	Rbank control	467.87	292.55
1	466.93	7.54	31	465.87	284.46
2	465.30	24.14	30	464.58	262.43
3	464.45	32.45	29	463.29	245.92
4	463.31	41.99	28	462.16	236.81
5	461.46	68.25	27	461.37	223.86
6	461.25	72.55	26	461.24	215.01
7	461.22	77.39	25	461.13	212.81
8	461.21	79.77	24	460.92	211.95
9	461.04	80.66	23	460.68	211.68
10	460.75	81.11	22	460.42	211.53
11	460.47	81.18	21	460.53	215.52
12	460.51	81.56	20	460.38	211.31
13	460.51	82.02	19	460.48	215.06
14	461.53	82.48	18	460.49	213.85
15	461.11	85.42	17	460.65	213.79
16	461.49	88.07	16	461.12	213.69
17	461.51	94.25	15	461.39	205.94
18	462.51	110.52	14	461.92	192.58
19	461.32	130.11	13	461.95	175.06
20	461.71	140.08	12	461.75	156.58
21	462.53	156.74	11	461.99	139.69
22	463.09	179.41	10	463.06	121.28
23	464.31	189.93	9	463.97	107.27
24	465.29	205.49	8	464.93	91.98
25	465.59	212.94	7	465.87	76.98
26	466.51	216.92	6	467.74	64.48
27	467.69	227.60	5	470.41	46.31
28	468.58	237.85	4	472.03	32.66
29	471.96	253.14	3	473.76	19.70
30	472.73	265.65	2	474.81	12.81
31	475.39	281.52	1	475.98	4.28
Lbank control	478.58	292.80	Lbank control	478.58	0.00

Cross section 2-1	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Rbank control	326.22	59.40	Rbank control	326.21	59.48
24	325.96	56.08	19	325.93	55.08
23	325.58	53.42	18	325.63	48.36
22	325.41	50.22	17	325.44	42.96
21	325.24	49.20	16	325.12	37.21
20	325.05	48.76	15	324.75	29.69
19	324.75	48.55	14	324.63	23.69
18	324.66	48.00	13	324.55	18.92
17	324.52	46.77	12	324.56	16.59
16	324.56	45.05	11	324.56	16.24
15	324.56	45.00	10	324.68	15.35
14	324.70	44.73	9	324.96	14.24
13	324.78	43.26	8	325.15	13.19
12	324.91	43.14	7	325.32	12.09
11	325.28	42.26	6	325.78	11.38
10	325.35	40.61	5	326.46	10.91
9	325.64	36.18	4	327.07	10.56
8	326.09	33.33	3	327.29	8.33
7	326.49	31.14	2	327.40	5.52
6	326.78	26.94	1	327.60	2.60
5	327.15	21.67	Lbank control	328.40	0.00
4	327.29	17.63			
3	327.39	13.28			
2	327.46	10.30			
1	327.75	5.25			
Lbank control	328.40	0.00			

Cross section 2-2	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Rbank control	323.66	59.34	Rbank Control	323.80	0.00
25	326.39	53.08	1	323.86	1.99
24	326.15	44.68	2	323.91	4.30
23	325.59	38.70	3	323.71	5.75
22	325.55	34.53	4	323.28	6.52
21	325.21	27.12	5	323.23	6.79
20	324.80	20.06	6	323.99	6.99
19	324.55	16.02	7	323.90	7.76
18	324.32	12.38	8	323.67	8.68
17	324.15	11.18	9	323.26	9.65
16	323.91	10.12	10	323.95	10.05
15	323.57	10.00	11	324.35	12.06
14	323.15	9.60	12	324.64	16.33
13	323.14	8.95	13	324.69	21.10
12	323.11	8.43	14	325.25	26.09
11	323.67	8.25	15	325.58	32.50
10	323.21	8.08	16	325.64	37.55
9	323.12	7.81	17	326.04	42.95
8	323.09	7.33	18	326.28	49.78
7	323.10	7.08	19	326.48	56.38
6	323.16	6.65	Lbank control	326.81	59.34
5	323.21	6.34			
4	323.47	5.57			
3	323.25	5.25			
2	323.55	5.12			
1	323.69	3.34			
Lbank control	326.93	0.00			

Appendix 14.3b **The height and number of each surveyed point for cross-sections 2-3, 3-1 and 4-1, and the distances between surveyed Points.**

Cross-section 2-3	2001			2002	
Point #	Height (MASL)	Distances (m)	Point #	Height (MASL)	Distances (m)
Blank control	322.83	0.00	Blank control	322.84	51.14
1	322.50	3.67	19	322.44	49.58
2	322.53	7.27	18	322.36	48.52
3	322.61	9.66	17	322.54	38.99
4	322.53	10.67	16	322.59	34.56
5	322.41	11.29	15	322.32	30.39
6	322.28	11.90	14	322.18	24.61
7	322.19	12.50	13	322.16	19.43
8	322.10	13.08	12	322.09	18.51
9	322.04	13.65	11	322.03	16.63
10	322.04	14.23	10	322.13	15.33
11	322.05	15.24	9	322.23	14.77
12	322.13	15.77	8	322.45	13.92
13	322.22	16.52	7	322.78	12.97
14	322.50	18.63	6	323.30	12.29
15	322.68	19.15	5	323.64	11.85
16	323.00	21.36	4	323.90	9.80
17	323.24	22.71	3	323.89	7.88
18	323.45	29.69	2	323.94	5.97
19	323.87	38.18	1	324.37	2.29
20	323.97	39.61	Blank control	324.41	0.19
21	323.88	44.76			
Blank control	324.41	51.17			

Cross-section 3-1	2001			2002	
Point #	Height (MASL)	Distances (m)	Point #	Height (MASL)	Distances (m)
Blank control	292.09	61.95	Blank control	282.79	0.00
29	291.65	60.36	1	281.71	2.96
28	290.62	59.02	2	281.93	4.12
27	290.67	56.17	3	280.89	5.96
26	279.30	50.90	4	280.57	5.71
25	279.13	47.48	5	280.04	7.36
24	278.58	42.15	6	279.32	8.47
23	278.30	38.27	7	279.12	9.28
22	278.00	35.73	8	278.44	9.95
21	277.80	29.36	9	278.28	11.12
20	277.73	27.85	10	277.65	12.03
19	277.73	25.64	11	277.78	12.37
18	277.67	22.20	12	277.73	12.57
17	277.74	19.16	13	277.69	13.08
16	277.70	15.82	14	277.69	13.77
15	277.96	15.40	15	272.69	14.28
14	278.10	14.77	16	277.71	14.45
13	278.60	14.12	17	277.78	15.23
12	278.98	13.82	18	278.18	15.64
11	278.88	12.96	19	278.38	16.29
10	278.90	12.60	20	278.67	18.64
9	278.78	12.25	21	278.60	21.07
8	278.95	11.97	22	278.68	25.36
7	280.23	10.90	23	278.84	27.97
6	281.83	9.73	24	278.83	31.19
5	283.73	9.11	25	279.57	33.94
4	284.75	8.31	26	280.32	38.03
3	286.92	7.01	27	281.29	39.89
2	292.20	5.51	28	282.33	42.87
1	285.12	2.82	29	283.15	45.02
Blank control	286.08	0.00	30	284.45	48.58
			31	285.32	52.27
			32	286.19	55.63
			33	286.17	60.31
			Blank control	286.03	61.02

Cross-section 4-1	2001			2002	
Point #	Height (MASL)	Distances (m)	Point #	Height (MASL)	Distances (m)
Blank control	541.62	0.00	Blank control	543.61	0.00
1	543.47	1.66	1	543.47	1.74
2	543.37	2.34	2	543.23	2.75
3	542.98	3.39	3	542.68	3.74
4	542.41	4.09	4	541.32	4.05
5	541.94	4.71	5	541.28	5.64
6	541.14	4.98	6	540.47	6.24
7	540.97	5.72	7	540.76	6.93
8	540.34	6.64	8	540.90	7.24
9	540.44	6.61	9	540.59	8.02
10	540.61	7.35	10	541.25	8.39
11	540.37	8.14	11	540.16	8.61
12	540.25	8.08	12	540.99	9.32
13	540.29	9.87	13	540.60	10.23
14	541.32	11.28	14	540.28	10.84
15	540.84	11.83	15	540.47	10.99
16	540.89	12.99	16	540.73	11.50
17	541.97	13.80	17	540.84	12.58
18	541.38	15.26	18	540.97	13.77
19	541.30	17.38	19	543.21	14.50
20	540.51	19.26	20	544.58	15.42
21	540.75	21.07	21	545.35	17.14
22	541.56	22.28	22	541.99	18.12
23	542.16	23.55	23	541.64	19.01
24	543.78	24.84	24	544.64	20.52
25	544.56	26.37	25	545.10	21.48
26	545.14	29.12	26	542.00	22.44
27	541.47	31.50	27	542.97	22.46
28	546.64	32.57	28	543.89	24.67
Blank control	546.65	41.76	29	544.64	26.26
			30	545.12	27.96
			31	545.42	29.57
			32	545.72	31.69
			33	545.80	32.15
			34	546.09	32.98
			35	546.67	35.23
			36	547.26	37.42
			37	547.72	41.08
			38	547.26	42.18
			Blank control	548.60	48.12

Appendix 14.3c The height and number of each surveyed point for cross-sections 4-2 and 4-3, and the distances between surveyed points.

Cross section 4-2	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Rbank control	143.10	0.00	Rbank control	143.09	0.00
1	142.97	1.71	1	142.92	1.98
2	142.52	3.23	2	142.31	4.03
3	142.16	4.52	3	141.99	4.85
4	141.54	5.41	4	141.46	5.90
5	141.32	5.96	5	140.43	6.88
6	141.18	6.27	6	139.95	7.84
7	140.48	6.71	7	139.85	8.29
8	140.12	7.14	8	139.58	8.74
9	139.79	7.65	9	139.41	8.96
10	139.32	8.25	10	139.45	9.15
11	139.35	8.62	11	139.6	9.45
12	139.49	9.75	12	139.64	9.78
13	139.69	10.72	13	139.74	10.21
14	140.05	11.27	14	139.94	11.46
15	140.58	12.79	15	140.47	12.58
16	142.61	13.37	16	140.52	13.89
17	142.41	14.96	17	141.39	14.55
18	143.18	15.79	18	140.17	15.84
19	142.04	16.73	19	140.06	17.01
20	143.18	17.80	20	140.23	18.02
21	143.24	19.58	21	140.57	18.63
22	140.92	20.32	22	140.3	18.89
23	141.66	21.93	23	140.32	19.73
24	142.68	24.18	24	140.88	20.49
25	143.61	26.47	25	141.81	22.27
26	143.88	27.77	26	142.61	24.04
27	144.59	30.29	27	143.56	26.30
28	144.85	33.09	28	143.9	28.35
29	146.79	41.65	29	144.6	30.92
Lbank control	146.79	41.86	30	144.82	32.94
			31	145.19	34.21
			32	145.66	37.68
			33	146.14	39.80
			Lbank control	146.79	41.93

Cross section 4-3	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Rbank control	142.55	0.00	Rbank control	142.55	0.00
1	142.36	1.60	1	142.38	1.19
2	142.08	2.81	2	142.09	2.77
3	141.90	3.87	3	141.27	4.46
4	141.58	4.57	4	141.23	6.07
5	140.81	5.16	5	141.09	7.16
6	140.20	6.25	6	139.79	10.60
7	139.75	9.24	7	140.64	12.83
8	139.55	11.17	8	139.9	13.84
9	139.98	12.07	9	139.24	14.09
10	139.85	13.10	10	139.09	15.10
11	139.78	13.86	11	138.58	15.25
12	139.51	14.12	12	138.54	15.92
13	139.12	14.32	13	138.82	16.84
14	139.01	14.75	14	139.09	17.16
15	138.79	15.38	15	139.17	17.20
16	139.09	16.07	16	139.45	17.48
17	139.03	16.71	17	139.89	17.93
18	139.03	17.07	18	140.51	18.37
19	139.04	17.07	19	141.61	19.00
20	139.41	17.42	20	141.57	20.28
21	139.91	17.81	21	143.27	22.00
22	140.61	18.43	22	143.89	23.95
23	141.75	18.81	23	144.77	25.91
24	141.60	19.54	24	145.32	29.51
25	141.83	20.10	Lbank control	145.62	30.16
26	141.28	21.97			
27	143.69	23.63			
28	144.65	25.39			
29	145.30	27.46			
Lbank control	145.62	30.14			

Appendix 14.3d **The height and number of each surveyed point for cross-sections 4-4, 4-5 and 5-1, and the distances between surveyed Points.**

Cross section 4-4	2001		2002		
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
R bank control	142.18	0.00	R bank control	142.13	35.59
1	141.99	1.18	32	142.08	34.68
2	141.69	3.10	31	141.84	32.88
3	141.48	4.30	30	141.04	30.33
4	141.03	5.15	29	139.88	27.52
5	140.61	5.83	28	138.57	26.86
6	140.00	6.58	27	137.77	26.72
7	139.55	7.20	26	137.79	26.32
8	138.97	7.67	25	137.75	25.34
9	138.55	8.01	24	138.00	25.15
10	138.59	8.24	23	138.68	24.25
11	138.48	8.45	22	138.84	23.08
12	138.25	8.65	21	138.59	21.77
13	137.82	8.85	20	138.61	20.12
14	137.43	9.48	19	138.96	19.24
15	138.06	10.23	18	138.42	18.73
16	138.10	10.88	17	138.86	18.61
17	138.24	11.36	16	138.01	18.00
18	138.55	12.21	15	138.08	16.80
19	138.52	13.46	14	138.23	14.76
20	138.60	14.59	13	138.64	13.05
21	138.65	15.41	12	137.84	12.50
22	139.41	16.29	11	137.93	11.99
23	139.70	17.49	10	137.95	11.82
24	139.76	19.65	9	139.70	11.63
25	139.76	21.76	8	139.28	11.52
26	140.03	24.88	7	139.65	11.03
27	141.17	27.43	6	140.05	10.56
28	142.48	29.06	5	140.84	9.68
29	143.33	29.59	4	142.66	6.15
30	144.12	31.14	3	143.62	5.05
31	144.64	33.19	2	144.38	3.53
L bank control	144.67	35.50	1	144.79	2.25
			L bank control	144.87	0.00

Cross section 4-5	2001		2002		
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
R bank control	141.73	0.00	R bank control	141.73	35.27
1	141.37	1.78	26	141.70	34.56
2	140.37	4.14	25	141.40	33.49
3	139.86	6.61	24	141.18	31.99
4	139.48	18.25	23	140.10	28.47
5	139.41	17.55	22	139.53	27.78
6	139.34	21.10	21	139.53	25.29
7	139.16	21.71	20	139.54	22.92
8	138.47	22.59	19	139.54	19.72
9	138.18	23.73	18	139.24	17.52
10	138.13	24.92	17	139.26	14.57
11	137.77	25.44	16	139.30	13.44
12	137.57	26.23	15	138.75	12.82
13	137.40	26.48	14	138.27	11.32
14	137.36	27.02	13	138.22	10.51
15	137.25	27.63	12	138.05	9.96
16	137.16	28.09	11	137.72	9.94
17	137.23	28.46	10	137.47	9.75
18	139.23	28.78	9	137.43	9.11
19	139.78	30.06	8	137.17	8.62
20	141.39	31.69	7	137.08	7.92
21	142.31	32.98	6	137.02	7.13
22	143.00	34.02	5	137.08	7.04
23	143.34	34.68	4	137.48	6.50
L bank control	143.32	35.79	3	139.10	4.60
			2	140.60	2.07
			1	142.79	1.10
			L bank control	143.32	0.00

Cross section 5-1	2001		2002		
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
R bank control	119.82	20.08	R bank control	118.87	19.95
20	118.80	17.59	20	118.82	19.04
19	118.63	16.09	19	118.80	17.05
18	118.27	14.95	18	116.58	15.81
17	118.00	13.85	17	117.39	13.83
16	117.47	12.65	16	116.65	11.67
15	116.66	11.72	15	116.20	11.13
14	115.87	11.27	14	115.92	10.46
13	115.75	10.33	13	115.84	10.07
12	115.43	9.88	12	115.59	9.74
11	115.54	9.72	11	115.48	9.20
10	115.41	9.05	10	115.41	8.11
9	115.34	8.54	9	115.42	7.93
8	115.38	7.71	8	115.44	6.89
7	115.42	6.63	7	115.55	6.58
6	115.54	6.31	6	115.55	6.57
5	116.24	5.80	5	115.68	6.27
4	116.42	5.14	4	116.06	5.94
3	116.42	3.67	3	116.43	4.99
2	116.65	2.18	2	116.56	3.04
1	117.03	1.37	1	116.96	1.57
L bank control	117.62	0.00	L bank control	117.62	0.00

Appendix 14.3e **The height and number of each surveyed point for cross-sections 5-2, 5-3, 6-1 and 6-2, and the distances between surveyed Points.**

Cross section 5-2	2001			2002	
Point #	Height (MAS)	Dist across (m)	Point #	Height (MAS)	Dist across (m)
Rbank control	115.23	22.14	Rbank control	116.23	22.17
25	117.96	23.81	23	117.90	20.54
24	117.4	19.25	22	116.95	18.43
23	116.76	17.8	21	116.45	16.93
22	116.25	16.89	20	115.88	15.05
21	116.03	15.81	19	115.24	13.74
20	115.79	14.82	18	115.00	12.50
19	115.42	14.07	17	114.85	11.96
18	115.03	13.01	16	114.62	11.45
17	114.81	12.2	15	114.48	11.23
16	114.55	11.3	14	114.40	11.06
15	114.38	10.52	13	114.36	10.74
14	114.33	10.35	12	114.28	10.42
13	114.23	9.59	11	114.22	9.51
12	114.24	8.65	10	114.15	8.64
11	114.21	8.85	9	114.20	8.25
10	114.32	8.36	8	114.33	8.20
9	114.48	7.76	7	114.42	8.03
8	114.52	7.5	6	114.49	7.62
7	114.59	7.1	5	114.64	7.11
6	114.57	6.71	4	114.91	6.23
5	114.99	6.09	3	115.35	5.25
4	115.19	5.43	2	115.63	4.53
3	115.63	4.77	1	115.83	2.45
2	115.7	3.48	Lbank control	115.96	0.00
1	115.92	1.19			
Lbank control	115.96	0			

Cross section 5-3	2001			2002	
Point #	Height (MAS)	Dist across (m)	Point #	Height (MAS)	Dist across (m)
Rbank control	117.83	0.00	Rbank control	117.83	23.70
1	117.40	1.85	21	117.80	22.60
2	116.67	3.61	20	117.22	21.57
3	116.40	4.75	19	116.53	19.49
4	115.97	5.78	18	115.80	17.91
5	115.56	7.37	17	115.51	16.38
6	115.33	8.81	16	115.35	14.69
7	115.21	9.90	15	115.16	13.90
8	115.11	9.97	14	115.01	13.47
9	114.92	10.43	13	114.90	13.29
10	114.63	10.58	12	114.31	13.08
11	113.53	11.01	11	113.94	12.69
12	113.93	11.31	10	113.94	12.15
13	113.85	11.69	9	113.93	10.91
14	113.71	12.21	8	113.99	10.09
15	113.70	12.76	7	114.06	9.82
16	113.72	13.43	6	114.10	9.55
17	113.81	14.00	5	114.54	8.49
18	114.04	14.45	4	114.72	7.12
19	114.37	15.32	3	115.18	5.78
20	114.53	16.15	2	115.40	3.29
21	114.85	16.95	1	115.65	1.70
22	115.20	18.56	Lbank control	115.81	0.00
23	115.43	20.23			
24	115.60	21.72			
Lbank control	115.81	23.71			

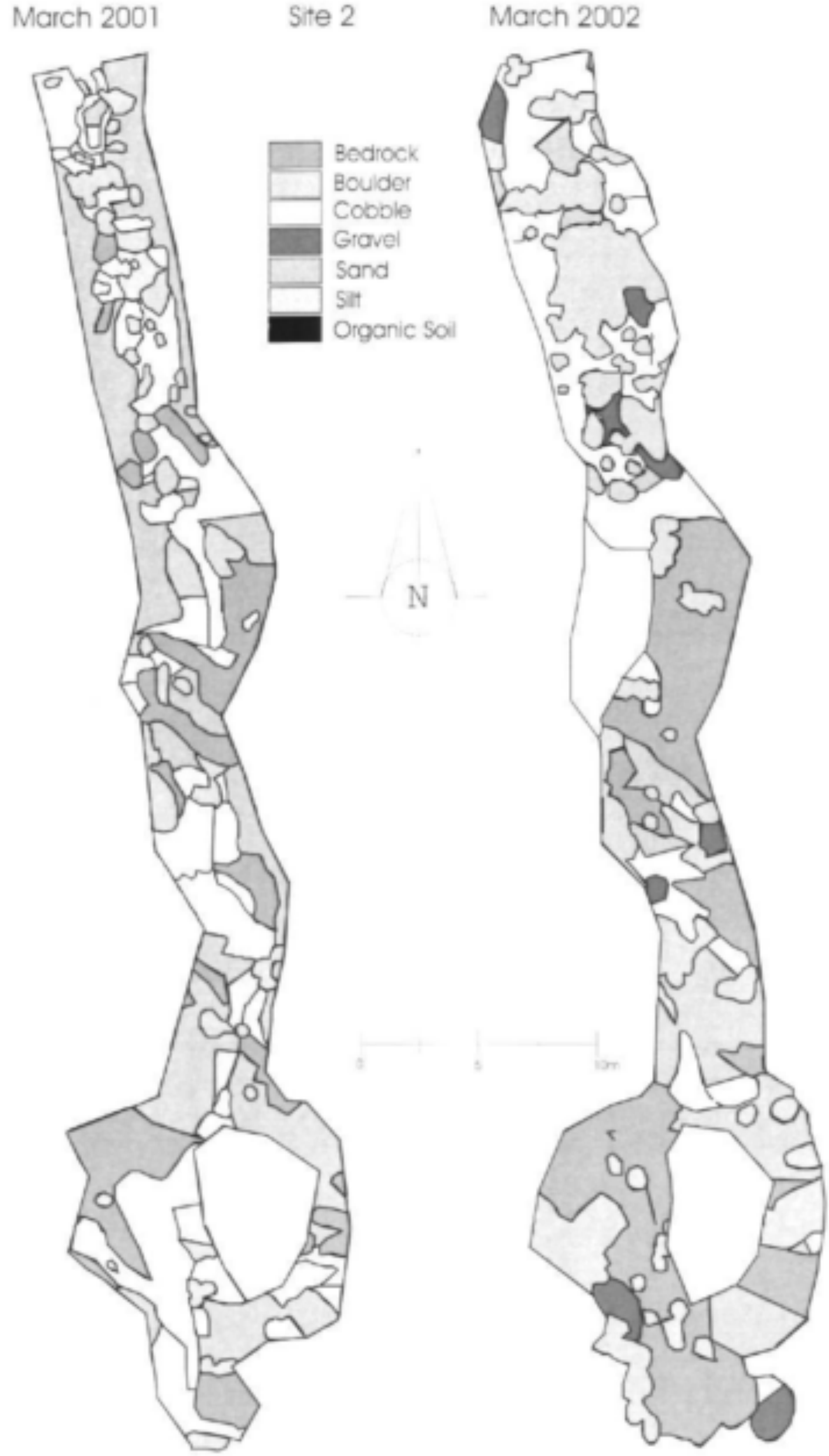
Cross section 6-1	2001			2002	
Point #	Height (MAS)	Dist across (m)	Point #	Height (MAS)	Dist across (m)
Rbank control	106.84	32.71	Rbank control	106.84	32.72
27	106.58	26.55	22	106.65	29.57
26	106.66	25.29	21	106.79	28.53
25	106.84	24.35	20	106.95	26.83
24	106.87	23.15	19	106.83	22.43
23	106.75	22.12	18	106.54	20.92
22	106.54	21.20	17	106.01	19.60
21	106.08	20.10	16	105.15	19.23
20	104.99	18.93	15	104.92	18.54
19	104.71	17.94	14	104.83	18.08
18	104.57	17.56	13	104.69	17.15
17	104.61	16.79	12	104.74	15.83
16	104.59	16.12	11	104.73	15.00
15	104.49	15.57	10	104.84	14.54
14	104.58	15.16	9	104.87	14.50
13	104.72	14.60	8	105.14	14.29
12	105.16	14.13	7	105.18	13.65
11	105.29	13.63	6	105.53	11.58
10	105.35	12.94	5	105.79	10.17
9	105.47	12.06	4	106.18	7.49
8	105.70	10.54	3	106.73	5.60
7	105.81	9.25	2	107.31	2.95
6	105.94	8.17	1	107.43	1.60
5	106.16	7.07	Lbank control	107.52	0.00
4	106.54	5.91			
3	106.94	4.95			
2	107.08	4.42			
1	107.29	2.30			
Lbank control	107.52	0.00			

Cross section 6-2	2001			2002	
Point #	Height (MAS)	Dist across (m)	Point #	Height (MAS)	Dist across (m)
Rbank control	106.00	0.00	Rbank control	106.30	21.06
1	105.68	2.14	16	105.64	18.97
2	105.61	3.35	15	105.31	16.77
3	105.10	5.16	14	104.97	14.55
4	104.90	6.64	13	104.51	13.30
5	104.45	7.77	12	104.30	12.96
6	104.31	8.27	11	104.26	12.88
7	104.18	8.97	10	104.16	12.44
8	104.14	10.23	9	104.21	10.75
9	104.12	10.95	8	104.18	9.57
10	104.06	12.16	7	104.13	8.45
11	104.14	12.97	6	104.16	8.11
12	104.59	14.05	5	104.25	7.96
13	104.93	14.50	4	104.55	7.19
14	105.57	16.24	3	105.10	5.53
15	105.81	17.48	2	105.63	4.00
16	106.23	19.21	1	106.22	2.13
Lbank control	106.39	21.05	Lbank control	106.39	0.00

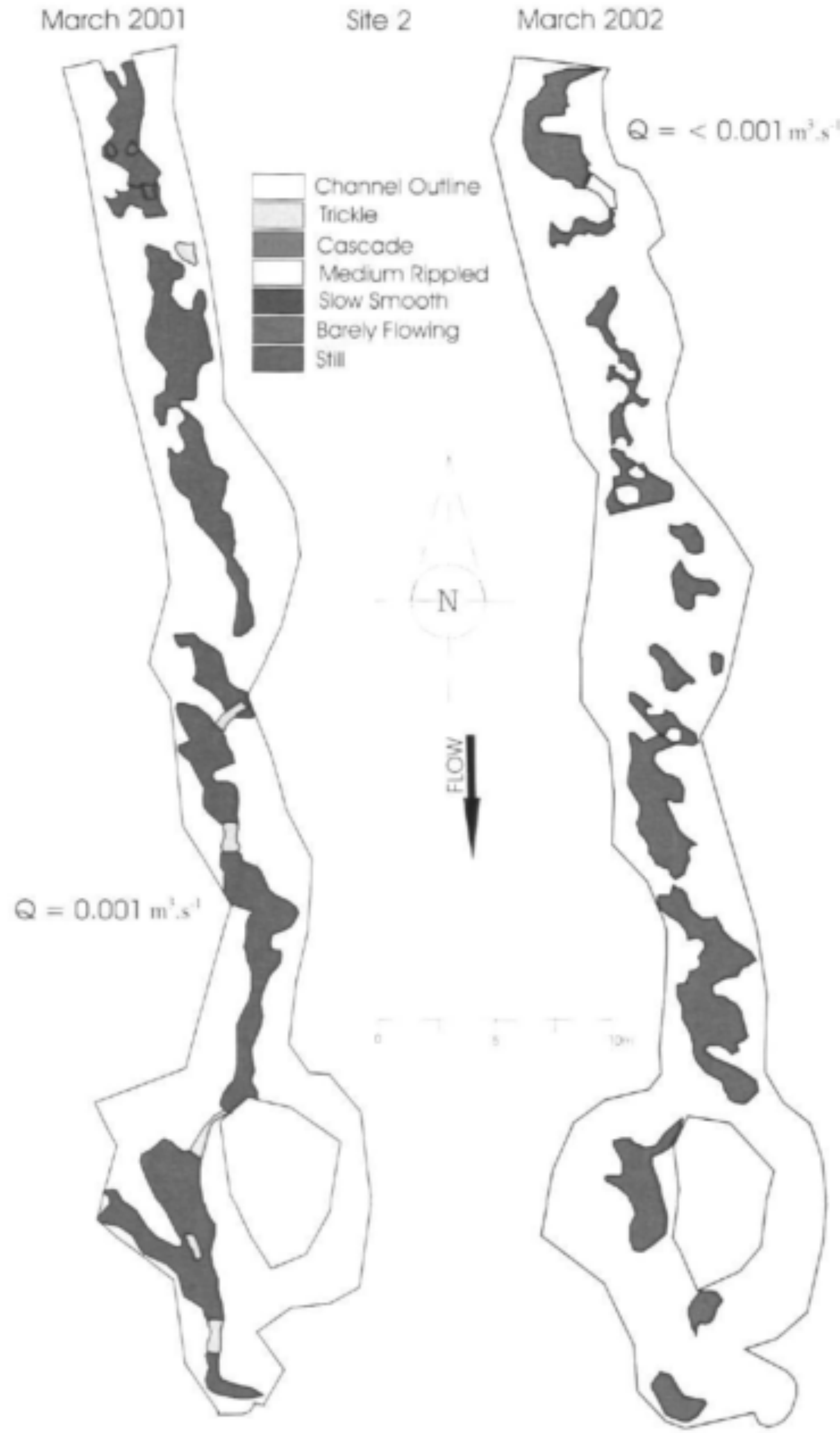
Appendix 14.3f The height and number of each surveyed point for cross-sections 1-1, 2-1 and 2-2, and the distances between surveyed Points.

Cross section 1-1	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Bank control	107.05	0.00	Bank control	107.09	51.33
1	106.72	3.62	18	106.78	47.02
2	106.47	5.06	17	106.84	42.82
3	106.03	8.43	16	106.00	37.54
4	105.64	12.14	15	105.36	31.60
5	104.83	16.24	14	104.26	26.05
6	104.54	17.86	13	103.52	24.79
7	104.16	18.47	12	103.19	23.01
8	103.55	19.72	11	103.07	22.21
9	103.43	20.53	10	102.81	21.98
10	102.96	21.18	9	102.17	21.65
11	102.23	21.88	8	102.04	20.24
12	102.49	22.17	7	102.63	19.75
13	102.33	22.36	6	102.20	19.37
14	102.33	23.34	5	103.38	17.09
15	102.33	24.12	4	103.18	14.96
16	103.18	25.58	3	103.49	7.72
17	103.20	27.09	2	103.94	4.60
18	103.17	29.04	1	103.63	2.14
19	103.45	34.10	Bank control	103.62	0.00
20	103.56	38.85			
21	103.88	43.13			
22	103.82	47.03			
Bank control	103.92	51.15			

Cross section 2-1	2001			2002	
Point #	Height (MASL)	Dist across (m)	Point #	Height (MASL)	Dist across (m)
Bank control	21.72	79.50	Bank control	21.70	79.56
35	21.85	78.38	30	21.84	77.54
34	21.82	77.44	29	21.59	75.95
33	21.68	76.50	28	21.16	74.62
32	21.47	75.71	27	20.87	74.08
31	21.36	75.09	26	20.68	72.08
30	20.96	74.09	25	20.79	71.75
29	20.96	73.81	24	20.69	70.79
28	20.79	72.17	23	20.67	69.63
27	20.61	72.25	22	20.56	68.97
26	20.51	71.44	21	20.67	67.59
25	20.42	69.72	20	20.76	67.53
24	20.38	68.35	19	20.74	67.14
23	20.52	67.20	18	21.19	66.52
22	20.91	66.66	17	21.48	65.15
21	21.23	65.87	16	21.50	63.70
20	21.37	65.25	15	21.80	61.76
19	21.30	63.69	14	22.22	59.26
18	21.50	62.67	13	22.63	55.17
17	22.35	57.79	12	22.47	49.53
16	22.63	51.68	11	22.55	45.39
15	22.81	48.52	10	23.02	38.86
14	22.21	46.79	9	22.85	33.16
13	22.62	46.32	8	22.71	28.88
12	22.65	41.81	7	23.48	26.11
11	23.02	37.76	6	23.48	25.55
10	23.44	36.09	5	25.45	22.11
9	23.87	35.19	4	26.67	18.08
8	23.43	34.34	3	26.05	12.57
7	22.85	33.32	2	26.52	7.70
6	22.98	31.06	1	27.46	3.53
5	23.55	26.70	Bank control	29.02	0.00
4	24.40	25.43			
3	25.59	21.32			
2	26.59	13.67			
1	28.06	10.57			
Bank control	29.01	0.00			



Appendix 14.4a **Substratum type map for Main Study Site 2.**



Appendix 14.4b **Flow type map for Main Study Site 2.**

Appendix 14.4c **Substratum type map for Main Study Site 5.**

Appendix 14.4d **Flow type map for Main Study Site 5.**

Appendix 14.5 Vegetation species checklist for specimens identified at all four Main Study Sites for both years. Alien plants are bracketed.

Species	Site 2		Site 4		Site 5		Site 6	
	2001	2002	2001	2002	2001	2002	2001	2002
[Acacia longifolia] (Andrews) Willd.			*	*	*	*		
[Acacia melanoxylon] R.Br.	*	*						
[Acacia saligna] (Labill.) H.L.Wendl.			*	*	*	*	*	
[Alstroemeria aurea] Graham								*
Apatesia pillansii N.E.Br.		*						
Asparagus aethiopicus L.								*
Asparagus declinatus L.	*							
[Avena barbata] Pott ex Link					*			
[Avena fatua] L.								
Berzelia lanuginosa (L.) Brongn.	*	*	*	*	*			
[Briza maxima] L.	*	*	*	*	*	*		*
[Briza minor] L.		*						
[Bromus diandrus] Roth		*						
[Bromus hordeaceus] L.		*						
Bromus leptoclados Nees	*							
Carpacoce spermacocea (Rchb.f.) Sond.	*	*						
Carpha glomerata (Thunb.) Nees			*	*	*	*		
Carpobrotus edulis (L.) L.Bolus	*	*				*		
Chenopodium album L.					*			
Chionanthus foveolatus (E.Mey.) Stearn							*	
Chrysanthemoides monilifera (L.) Norl. ssp. *monilifera*		*						*
Chrysocoma coma-aurea L.			*	*				
Cliffortia dodecandra Weim.		*						
Cliffortia subsetacea (Eckl. & Zeyh.) Diels ex Bolus & Wolley-Dod		*						
[Conyza bonariensis] (L.) Cronquist				*				
[Conyza canadensis] (L.) Cronquist	*	*	*	*	*	*		*
Cotula turbinata L.		*						
Cotula vulgaris Levyns		*						
[Cucurbita moschata] (Duch. ex Lam.) Duch. ex Poir.	*							
Cunonia capensis L.	*	*						
Cuscuta nitida E.Mey. ex Choisy				*		*		
Cynodon dactylon (L.) Pers.	*							
Cyperus species		*		*		*		
Digitaria debilis (Desf.) Willd.	*	*		*	*	*		*
Disparago anomala Schltr. ex Levyns			*					
Droguetia ambigua Wedd.								*
Ehrharta calycina Sm.			*		*		*	*
Ehrharta erecta Lam.						*		
Ehrharta ramosa (Thunb.) Thunb. ssp. *aphylla* (Schrad.) Gibbs-Russ.		*		*				
Ehrharta setacea Nees ssp. *uniflora* (Burch. ex Stapf) Gibbs-Russ.	*	*	*	*	*	*		
Elegia thyrsifera (Rottb.) Pers.		*						
Erica caffra L.			*					
Erica hirtiflora Curtis			*	*				
Erica laeta Bartl.		*						
[Eucalyptus grandis] W.Hill ex Maiden			*	*				
Ficinia brevifolia Nees ex Kunth		*	*					
Ficinia bulbosa (L.) Nees		*						
Ficinia filiformis (Lam.) Schrad.	*			*				
Ficinia indica (Lam.) Pfeiff.		*	*	*				
Ficinia oligantha (Steud.) J.Raynal	*		*					
Ficinia secunda (Vahl) Kunth		*	*	*				
Ficinia species				*				
Ficinia tenuifolia Kunth			*					
Fuirena hirsuta (P.J.Bergius) P.L.Forbes	*		*	*	*			

Appendix 14.5 Vegetation species checklist for specimens identified at all four Main Study Sites for both years. Alien plants are bracketed.

Species	Site 2		Site 4		Site 5		Site 6	
	2001	2002	2001	2002	2001	2002	2001	2002
[Fumaria muralis] Sond. ex W.D.J.Koch								*
Helichrysum crispum (L.) D.Don			*					
Helichrysum cymosum (L.) D.Don	*	*	*	*	*	*		
Helichrysum indicum (L.) Grierson		*						
Helichrysum litorale Bolus		*						
Helichrysum pandurifolium Schrank								*
Hellmuthia membranacea (Thunb.) R.W.Haines & Lye	*		*	*				
Histiopteris incisa (Thunb.) J.Sm.	*	*	*	*	*	*		
Indigofera capillaris Thunb.	*							
Ischyrolepis tenuissima (Kunth) H.P.Linder	*	*						
Isolepis ludwigii (Steud.) Kunth	*	*						
Isolepis marginata (Thunb.) A.Dietr.	*	*	*	*	*	*		
Isolepis prolifer R.Br.	*	*	*	*	*	*	*	*
Isolepis tenuissima (Nees) Kunth	*							
Juncus capensis Thunb.	*	*	*	*	*	*		*
Juncus effusus L.		*		*		*		*
Kiggelaria africana L.							*	*
[Lagurus ovatus] L.	*	*						*
Lampranthus species		*						
[Lantana camara] L.					*	*	*	*
Laurembergia repens P.J.Bergius	*	*	*	*	*	*		
Laurentia secunda (L.f.) Kuntze	*	*						
Leonotis leonurus (L.) R.Br.						*		
Lobelia comosa L.			*	*				
Lobelia erinus L.				*				
Metalasia muraltiifolia DC.								
Metalasia muricata (L.) D.Don	*	*	*	*				
Micranthus alopecuroides (L.) Rothm.	*	*						
Oftia africana (L.) Bocq.		*	*	*		*		
Otholobium parviflorum (E.Mey.) C.H.Stirt.			*					
Othonna parviflora P.J.Bergius			*	*				
Othonna quinquedentata Thunb.			*					
[Paraserianthes lophantha] (Willd.) I.C.Nielsen			*		*		*	*
[Paspalum dilatatum] Poir.						*		
[Paspalum urvillei] Steud.	*	*	*	*	*	*		
Passerina vulgaris Thoday			*	*				
Pelargonium alchemilloides (L.) L'Hér.				*				
Pelargonium chamaedryfolium Jacq.	*		*					
Pelargonium cucullatum (L.) L'Hér. ssp. cucullatum	*	*	*	*	*	*		
[Pennisetum clandestinum] Chiov.	*	*				*		*
Pennisetum glaucocladum Stapf & C.E.Hubb.						*		
Pentaschistis airoides (Nees) Stapf	*		*	*				
Pentaschistis curvifolia (Schrad.) Stapf		*						
Pentaschistis glandulosa (Schrad.) H.P.Linder			*	*		*		
Pentaschistis pallida (Thunb.) H.P.Linder		*						
[Persicaria serrulata] (Lag.) Webb & Moq.							*	
[Physalis peruviana] L.								*
[Phytolacca americana] L.			*	*	*	*		*
[Pinus pinaster] Aiton	*	*						
Polypogon strictus Nees					*	*		*
[Populus x canescens] (Aiton) Sm.							*	*
Prionium serratum (L.f.) Drège ex E.Mey.			*	*				
Prismatocarpus sessilis Eckl. ex A.DC.	*	*						
[Pseudognaphalium luteo-album] (L.) Hilliard & B.L.Burtt		*						

Appendix 14.5 **Vegetation species checklist for specimens identified at all four Main Study Sites for both years.** Alein plants are bracketed.

Species	Site 2 2001	Site 2 2002	Site 4 2001	Site 4 2002	Site 5 2001	Site 5 2002	Site 6 2001	Site 6 2002
Pseudognaphalium undulatum (L.) Hilliard & B.L.Burtt				*				
Pseudoselago serrata (P.J.Bergius) Hilliard			*					
Psoralea pinnata L.	*	*	*	*				
Psoralea restioides Eckl. & Zeyh.		*						
Pteridium aquilinum (L.) Kuhn	*	*	*	*	*	*		*
Pycreus polystachyos (Rottb.) P.Beauv.	*	*		*		*		*
[Quercus robur] L.		*						
Rhus lucida L.	*	*	*	*				
[Ricinus communis] L.								*
[Roella ciliata] L.			*					
Rorippa fluviatilis (E.Mey. ex Sond.) Thell. var. caledonica (Sond.) Marais								*
[Rubus pinnatus] Willd.					*	*		
[Rumex acetosella] L.		*			*	*		
Senecio arenarius Thunb.				*				
Senecio burchellii DC.				*	*	*		*
Senecio crassiusculus DC.					*			
Senecio elegans L.								*
Senecio pinnulatus Thunb.		*	*					
Senecio pterophorus DC.								*
Senecio pubigerus DC.	*	*	*	*	*	*		
Senecio rigidus L.	*	*	*	*	*	*		
Solanum nigrum L.	*	*	*	*		*		*
[Sonchus oleraceus] L.			*					*
Sporobolus africanus (Poir.) Robyns & Tournay	*	*	*	*				
Stoebe cinerea (L.) Thunb.								
Stoebe fusca (L.) Thunb.		*						
Syncarpha vestita (L.) B.Nord.		*						
[Taraxacum officinale] Weber sensu lato	*	*		*	*	*		*
Tetraria capillacea (Thunb.) C.B.Clarke			*					
Tetraria exilis Levyns		*						
Themeda triandra Forssk.				*				
Todea barbara (L.) T.Moore	*	*	*	*				
Trachyandra divaricata (Jacq.) Kunth				*				
Ursinia anthemoides (L.) Poir.			*					
Ursinia tenuifolia (L.) Poir.		*						
Vellereophyton dealbatum (Thunb.) Hilliard & B.L.Burtt			*					
Virgilia oroboides (P.J.Bergius) Salter	*	*						
Wahlenbergia parvifolia (P.J.Bergius) Lammers			*	*				
Zantedeschia aethiopica (L.) Spreng.	*	*						*

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Development of river rehabilitation in Australia: Lessons for South Africa

AC Uys

The aim of this WRC report is to convey, at a reasonably non-technical level, what South Africa can learn from the decade of development of river rehabilitation in Australia, and from their recently-published national-level guidelines and methodologies. This report is addressed to South African and Australian readers from diverse backgrounds, and has thus been aimed at a non-technical level. In addition, geographic, environmental and socio-political 'context' has been included for Australia, as this provides the background to what drove the initiation and development of river rehabilitation. Where this is not of interest to the more scientific reader, it is advised that these individuals skip to chapters of a more technical nature. The introductory and concluding chapters are of relevance to all. The introductory chapter provides a background to the field of rehabilitation, and a discussion of the many related terms in use to describe this discipline. While both the terms 'restoration' and 'rehabilitation' are commonly used, the more academic interpretation is that the term 'restoration' implies a return to natural pre-impact state and is thus aspirational and seldom achievable, while rehabilitation focusses on achievable objectives and also aims for improvement and protection, with the aim of the system eventually resembling its pre-impact state. The aim of 'remediation' is to improve the ecological condition of the river, while not aiming for an endpoint that resembles its original condition (Breen and Walsh in Rutherfurd and Fryirs, 1999). **On the basis that, at least in contemporary literature, equal meaning is generally attached to the use of the terms 'restoration' and 'rehabilitation', these are used interchangeably in this report.**

The issue of system 'recovery' is fundamental to rehabilitation. Recovery relates to the ways in which systems respond to disturbances, both in a geomorphic and biological sense. One of the difficulties in rehabilitation planning lies in assessing whether a system is in a state of recovery, and if so, where along the recovery trajectory it lies.

South Africa is well placed to absorb the lessons of international experience and to become world class in the field of urban and non-urban river restoration without having to reinvent the wheel. Failure to recognise the timeliness of linking to, or learning from, external initiatives in river rehabilitation and protection is likely to prove extremely expensive both in economic and in environmental terms. It is less a matter of 'what are the costs of doing it', and more a matter of ***'what are the costs of not doing it?'***

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