TRACKING MOVEMENT OF LARGE FISH SPECIES THROUGH A RIVER SYSTEM: METHODS DEVELOPMENT

By

Bruce R Paxton

Zoology Department
University of Cape Town
Private Bag
Rondebosch
7701

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PROJECT STAFF

Dr P-A Scherman (IWR *)
* currently of Coastal & Environmental Services, PO Box 934, Grahamstown 6140

Dr S Jooste (RQS, DWAF)

Dr WJ Muller (IWR)

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EXECUTIVE SUMMARY

During 1998 the Water Research Commission (WRC) funded a project to investigate the feasibility of using a risk-based approach for setting Resource Quality Objectives (RQO) for the protection of water resources (Jooste *et al.*, 2000). The investigation intended to provide direction for future, more detailed research in order to support, develop and implement the protection-based classification system for water resources, as undertaken by the Department of Water Affairs and Forestry (DWAF) in order to meet the objectives of the National Water Policy (DWAF, 1997) and National Water Act (No. 36 of 1998). This project formed Phase I of the risk-based objectives study, with the findings of the current report being Phase II. Both Phases I and II attempted to investigate and evaluate risk-based methodologies, which could effectively incorporate the uncertainty and variability inherent to biological systems and biological data. While Phase I reviewed research into setting objectives for water quantity, water quality, habitat integrity and biotic integrity requirements of water resources; investigated possible approaches to incorporate risk concepts into the protection of water resources, and identified future research directions; Phase II focussed on linking stressors and responses to an identifiable end-point, and integrating stressor responses for co-occurring stressors.

The primary focus of Phase II was therefore a specialist workshop held in February 2000, with the overall aim of linking stressors and associated responses to an identified end-point, and integrating stressor responses for co-occurring stressors. The stressors considered were changes to flow (water quantity), water quality and habitat. The following discussion points were identified and defined to lead the workshop:

- If the aim is to integrate stressor effects, a common end-point (e.g. the sustainability requirements of the Ecological Reserve) must be defined.
- How should stressors be defined so that a given response can be linked to the stressor stimulus, and can the response be related in some way to the identified end-point?
- What techniques, procedures or protocols already exist to determine stressor exposure and effects?
- Can risk-based objectives (quantitative or semi-quantitative) reasonably be derived for use within the present regulatory framework?

Outcomes of discussions surrounding these identified questions are detailed in Sections 3.2 to 3.5 of the report, with Chapter 4 listing identified gaps and recommendations.

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GLOSSARY

This glossary of terms is amended from Jooste *et al.* (2000).

Acute effect (exposure) value	The concentration at and above which statistically significant acute adverse effects are expected to occur (DWAF, 1996).
Analysis	A formal, usually quantitative, determination of the effects of an action (as in risk analysis and impacts analysis) (Suter, 1993).
Assessment	The combination of analysis with policy-related activities such as identification of issues and comparison of risks and benefits (as in risk assessments and impacts assessment) (Suter, 1993).
Biodiversity	The diversity of living things found in the natural world. The concept usually refers to the different species, but also includes ecosystems and the genetic diversity within a given species (Bush, 1997).
Criterion	The level of exposure (concentration and duration) of a contaminant in a particular medium that is thought to result in an acceptably low level of effect on populations, communities, or uses of the medium (e.g. water quality criteria, air quality criteria) (Suter, 1993).
Chronic effect (exposure) value	The concentration limit, which is safe for all or most populations even during continuous exposure (DWAF, 1996).
Deterministic analysis	An analysis in which all population and environmental parameters are assumed to be constant and accurately specified (Suter, 1993).
Ecological integrity	The ability of an ecosystem to support and maintain a balanced, integrated composition of physico-chemical habitat characteristics, as well as biotic components, on a temporal and spatial scale, that are comparable to the natural (i.e. unimpaired) characteristics of such an ecosystem. High ecological integrity implies that the structure and functioning of an ecosystem are unimpaired by anthropogenic stresses) (Murray, 1999).
Ecological risk analysis	Determination of the likelihood (usually expressed as probability) and magnitude of adverse effects of environmental hazards (chemical, physical, or biological agents occurring in or mediated by the ambient environment) on nonhuman biota (Suter, 1993; Jooste, 2001).
Ecological risk assessment	The process of defining and quantifying risk to nonhuman biota and determining the acceptability of those risks (Suter, 1993).

Ecological Water Requirements — Ecological Reserve

Ecosystem — A biotic community and its interaction with the abiotic environment (Bush, 1997).

Effects assessment — The component of an environment risk analysis that is concerned with quantifying the manner in which the frequency and intensity of effects increase with increasing exposure to a contaminant or other source of stress (Suter, 1993).

Endpoint, assessment — A quantitative or quantifiable expression of the environmental value considered to be at risk in a risk analysis, e.g. a 25% reduction of a particular species (Suter, 1993).

Environmental risk analysis — Determination of the probability of adverse effects on humans and nonhuman biota resulting from an environmental hazard (a chemical, physical or biological agent occurring in or mediated by the environment) (Suter, 1993).

Hazard — A state that may result in an undesired event, the cause of risk (Suter, 1993).

Hazard assessment — Determination of the existence of a hazard.

(a) In predictive risk assessments, it is a preliminary activity that helps to define assessment endpoints by determining which environmental components are potentially exposed to toxic concentrations and how they might be affected.

(b) An alternate assessment method that determines whether a hazard exists by comparing the magnitude of expected environmental concentrations to toxicological test endpoints for a contaminant (Suter, 1993).

Instream Flow Requirements — Some flows within a total flow regime in a river are more important than others for maintenance of the river ecosystem. These flows can be identified and described in terms of their timing, duration and magnitude. These identified flows can be combined to define a recommended modified flow regime specific for that river and constitutes the instream flow requirement (King and Louw, 1998).

Mesocosm — Medium-sized multi-species system in which physical and biological parameters can be altered and subsequent effects monitored. They may be field- or laboratory-based and are thought to mimic responses of organisms in the field more realistically than single-species test systems (Palmer and Scherman, 2000).

Model — A formal representation of some component of the world. Models may be mathematical, physical or conceptual (Suter, 1993).

Parameter uncertainty The component of uncertainty associated with estimating model parameters. It may also arise from measurements or extrapolation (Suter, 1993).

Reserve The quantity and quality of water required -
(a) to satisfy basic human needs by securing a basic water supply, as prescribed under the Water Services Act, 1997 (Act No. 108 of 1997), for people who are now or who will, in the reasonably near future, be -
  (i) relying upon;
  (ii) taking water from; or
  (iii) being supplied from,
  the relevant water resource; and
(b) to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource (National Water Act, No. 36 of 1998).

Resilience (ecological) Resilience measures the rate of return to a predisturbance state after a perturbation, and is directly related to ecosystem recovery (Suter, 1993).

Resource A water resource includes a watercourse, surface water, estuary or aquifer (National Water Act, No. 36 of 1998).

Resource base The base level of ecological integrity and function which must be maintained in order to protect the ecological resilience of a water resource, so that the capability of the resource to supply services or meet the needs of humans can be maintained in the long term (National Water Act, No. 36 of 1998 - Part 1, Section 1.3.4).

Resource quality The quality of all the aspects of a water resource including
(a) the quantity, pattern, timing, water level and assurance of in-stream flow;
(b) the water quality, including the physical, chemical and biological characteristics of the water;
(c) the character and condition of the in-stream and riparian habitat; and
(d) the characteristics, condition and distribution of the aquatic biota (National Water Act, No. 36 of 1998).

Resource Quality Objective A numerical or descriptive statement of the conditions, which should be met in the receiving water resource to ensure that the resource is protected (National Water Act, No. 36 of 1998 - Part 1, Section 1.2.4).

Risk The likelihood (usually expressed as probability) of a prescribed undesired effect. If the level of effect is treated as a number, risk is a function of the likelihood and frequency of effect. Risk results from the existence of a hazard and uncertainty about its expression (Suter, 1993).

Risk assessment	The process of assigning magnitude and likelihood to the adverse effects of human activities or natural catastrophes (Suter, 1993).
Risk characterization	The process of (a) integrating the exposure and effects assessments to estimate risks and (b) summarizing and describing the results of a risk analysis for a risk manager or other stakeholders (Suter, 1993).
Risk management	The process of deciding what actions to take in response to a risk (Suter, 1993).
Stochastic	Randomly determined; that follows some random probability distribution or pattern so that its behaviour may be analysed statistically but not predicted precisely (Brown, 1993); that which cannot be determined uniquely, but can only be expressed in terms of likelihood.
Stress	The proximate cause of an adverse effect on an organism or system (Suter, 1993).
Stressor	Any physical, chemical or biological entity or process that can induce an adverse response (Murray and Claassen, 1999).
Sustainability (ecological)	The need to maintain ecological structures, functions or ecological integrity (Simonovic, 1996).
Toxicity	(1) The harmful effects produced by exposure of an organism to a chemical; (2) The property of a chemical that causes harmful effects in organisms (Suter, 1993).
Uncertainty	Imperfect knowledge concerning the present or future state of the system under consideration; a component of risk resulting from imperfect knowledge of the degree of hazard or of its spatial and temporal pattern of expression (Suter, 1993).
Xenobiotic	A toxicant or foreign substance (Rand, 1995).

LIST OF ACRONYMS

This list of abbreviations is amended from Jooste *et al.* (2000).

AEV	Acute Effect (Exposure) Value
BAT	Best Available Technology
BATNEEC	Best Available Technology Not Exceeding Excessive Cost
CAP	Continuous Assessment Paradigm
CEV	Chronic Effect (Exposure) Value
CSIR	Council for Scientific and Industrial Research
CV	Criterion Value
DEAT	Department of Environmental Affairs and Tourism
DRIFT	Downstream Response to Imposed Flow Transformations
DSS	Decision Support System
DWAF	Department of Water Affairs and Forestry
EC	Environmental Concentration
EIA	Environmental Impact Assessment
EMC	Ecological Management Class
EMP	Environmental Management Plan
ERA	Ecological Risk Assessment
ERBM	Ecological Risk-Based Management
ECR	Ecological Reserve Category
FS-R	Flow Stressor-Response
I&APs	Interested and Affected Parties
IEM	Integrated Environmental Management
IFR	Instream Flow Requirement
IWQS	Institute for Water Quality Studies
IWR	Institute for Water Research
FDC	Flow duration curve
LC50	Concentration that kills 50% of the test population
LOEC	Lowest Observed Effect Concentration
LT50	Lethal time
NER	No Effect Range
QAP	Quantal Assessment Paradigm
PEC	Predicted Environmental Concentration
PNEC	Predicted No observed Effect Concentration
RBO	Risk-based objectives
RQO	Resource Quality Objectives
RQS	Resource Quality Services
RU	Rhodes University
SSR	Stressor Response Relationship
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
TWQR	Target Water Quality Range
UCT	University of Cape Town
UCEWQ	Unilever Centre for Environmental Water Quality
US EPA	United States Environmental Protection Agency

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CHAPTER 1

THE RISK ASSESSMENT FRAMEWORK IN SOUTH AFRICA

1.1 INTRODUCTION

The fundamental differences between hazard and risk-based approaches can be described as follows: A hazard-based approach emphasizes the *potential* for causing an effect, while risk-based approaches define a realistic *expectation* of effect. It is important to understand the difference between these approaches before a risk-based approach to resource management can be undertaken.

A risk-based approach generally refers to a process whereby a desired outcome is set as management objective relating to specific risk levels, and all stressors on a system are managed so as to achieve these objectives (Claassen and Wade, 2001). The cumulative impact of stressors is therefore managed. This approach presents a shift from present management strategy, where individual stressors are normally managed independently of their combined impacts.

The incorporation of risk concepts into water resource management is still in its infancy in South Africa. The concept of Ecological Risk Assessment (ERA), and its application to informed decision-making, is embodied in a text by Claassen *et al.* (2001). This document provides a background to the ERA process, and provides a set of guidelines by which an ERA can be undertaken. Although this Water Research Commission-funded study has demonstrated the usefulness of risk-based procedures, the Department of Water Affairs and Forestry (DWAF) is still drafting its risk policy document. It is assumed that the availability of such a policy document, which is envisaged to cover concepts such as risk-based decision-making and building capacity within the field of risk assessment, will further clarify the role of risk concepts in water resources management in South Africa.

The South African National Water Act (No. 36 of 1998) recognizes that the water resource needs to be used to the benefit of the population, but also needs to be protected. Water use (including the discharge of wastes) is necessary for the economic development of the country and its people, while protection of the resource is needed so that this use as well as the other goods and services provided by the resource, will be available to future generations. Therefore, risk and risk-based management is conceptually well-suited to resource management in this country. Risk not only takes cognisance of the potential effect of human activity, but also considers the environment in which this activity takes place. It is able to provide an objective, focussed evaluation of impact that, if used properly, provides the basis for balanced management decision. Risk-based management of the water resource has been used in water quantity supply management for many years by DWAF. The application of risk-based management now needs to be extended to water quality management.

Furthermore, the concept of risk is incorporated in the approach followed, and methods and tools used, for determining the Ecological Water Requirements (EWR) for aquatic water resources. This is fundamental to the process as the Ecological Reserve concerns aquatic ecosystems, which are inherently variable. Some of the methods used, e.g. the stressor-response approach used for flow determinations, incorporate risk concepts more than others, e.g. the approach used for water quality assessments, which is still largely hazard-based. All EWR methods are however moving toward an effective incorporation of risk concepts.

The development of a classification system for the country's rivers is still being undertaken, and will be based on the risk of irreversible damage to the resource base. The concept is therefore risk-based, but again methods incorporating risk must be developed or present methods refined.

This chapter of the report therefore serves to describe and define risk-based methods, and provide some background to the current status and use of these methods.

1.2 ECOLOGICAL RISK ASSESSMENT: AN OVERVIEW

This summary is based largely on the work of Claassen and colleagues, as described in their document published in June 2001, and attempts to capture the main points of ERA in South Africa.

To achieve effective resource management and decision-making, often in the absence of adequate data, an approach is needed which can incorporate the uncertainty and variability inherent to biological data.

Basic characteristics of risk are as follows (Jooste and Claassen, 2000):

- An **expression of likelihood**. This may be in the form of probability or possibility.
- A **subject** (hazard or stressor) which initiates the consideration of risk.
- An **object** (target) upon which the stressor or hazard is expected to have an effect.
- The **type** (or consequence) and **magnitude** of effect of impact being assessed (i.e. the probability of something happening to the object).

Differences between risk and hazard can therefore be expressed as follows (Jooste and Claassen, 2000). **Risk** is the likelihood or expectation of a specified effect and incorporates both the relationship between the stressor and its expected response, as well as the expectation of the exposure to that stressor. It accepts that the some relationship exists between the stimulus applied to a system and the response of that system, and that this response is practically continuous above a possible threshold stimulus. Uncertainty and variability becomes part of the expression of risk. Risk as an assessment tool is therefore used most effectively in a risk management framework.

Hazard focuses on the potential of a stressor to cause an effect which is seldom explicitly specified. It is not explicitly concerned with the uncertainty and variability within the effects on the system under consideration. The concept of a criterion is the focus of this approach. The outcome of the assessment is only concerned with the confidence with which compliance to the criterion can be expressed. The assessment result is therefore essentially dichotomous or binary (e.g. complies / does not comply).

In 1997, Skivington produced a guide for assessing risk from point sources of pollution. Concurrently, the United States Environmental Protection Agency (US EPA) developed guidelines for the application of ERA in America. These guidelines were published in 1998. The US EPA approach formed the basis for the method developed by Murray and Claassen (1999) for use within South Africa.

An ERA therefore determines the likelihood that undesirable ecological effects may occur, or are occurring, as a result of exposure to one or more stressors. The approach developed by

Murray and Claassen (1999) incorporates the following abbreviated steps (Claassen and Wade, 2001):

- **Agree on objectives**: this phase requires extensive liaison and agreement regarding objectives, acceptable levels of uncertainty, goals, and resources (financial, human and data) availability.

- **Formulate analysis plan**: Technical data gathering e.g. stressor sources and potential impacts, system at risk, assessment end-points, and possible exposure routes. The risk hypotheses are therefore developed at this point.

- **Analyse information**: Quantify exposure and effects characteristics, and identify data uncertainty and variability. Collect additional data if required, and address all aspects of the stressor(s). Prepare a stressor-response profile, which generates the following information.

    - Stressor level – represents any stressor that may have an adverse effect on the end-point. (Alternative terminology for *end-point* is *target population* or *receptor*. The latter terms are used widely in the US and UK as the biotic entity involved is the end-point (Jooste, RQS, pers. comm.)).
    - Exposure profile – indicates the probability of the end-point being exposed to different levels of the stressor.
    - Effects profile – shows a cumulative distribution of effects on the end-point, given specific exposures.
    - Risk profile – quantifies the co-occurrence of the exposure and effects.

    A number of tools are available for use during the analysis phase of an ERA. Examples of *risk assessment models* include the following (DEAT, 2000):

    - APPRAISE: database and calculation tool to assess the environmental impact of industrial releases (UK).
    - RBCA: Risk-based Corrective Action Tool Kit for contaminated land and water (UK).
    - REFEREE: ERA using effect models linked to ecological and ecotoxicological databases (Netherlands).
    - CalTOX: A multimedia total exposure model for hazardous waste sites (USA).

    Examples of *fate and transport models* (used to determine the effective dose with which the end-point (e.g. ecosystem) will be in contact) are the following (DEAT, 2000):

    - AQUA: Groundwater flow and contaminant transport mode (USA).
    - PLUMES: Dilution / dispersion model for pollution plumes in marine and freshwater (USA).
    - WASP: Water Quality Analysis Simulation Programme models; contaminant fate and transport in surface waters (USA).

    Effects can be determined using the following tools (DEAT, 2000):

    - Pulsed exposures.

- Population models.
- Sensitivity distributions.
- Sediment toxicity evaluations.
- Chronic toxicity tests.
- Mesocosms and microcosms.
- Behavioural toxicity tests.

A number of databases are available for use, e.g. the International Register of Potentially Toxic Chemicals (UNEP), Integrated Risk Information System (US EPA) and Ecotox Thresholds Software (US EPA).

- **Characterise and communicate risk**: The likelihood of adverse ecological effects is determined by integrating exposure and effect data, and evaluating associated variability and uncertainties. Risk hypotheses are tested, and the results of the risk assessment are presented in the appropriate manner. Risk hypotheses are therefore predictions of relationships between stressor, exposure and the response of the assessment end-points.

- **Manage risk**: Results are discussed with the risk manager, and decision-making commences. An iterative approach to risk assessment may be followed.

Fundamental to risk assessment is therefore the hazard or stressor that is eliciting the risk (and its associated exposure pathway), and the probability of effect due to that exposure, i.e. the stressor-response relationship.

1.3 RISK AND RESOURCE MANAGEMENT

The concepts of hazard and risk are used extensively in resource management, particularly water resource management. Examples of how hazard and risk can be incorporated into environmental management and water resource management are shown in Sections 1.3.1 and 1.3.2 respectively.

1.3.1 Integrated Environmental Management (IEM)

Risk-based approaches can be used for a wide number of applications, including environmental studies such as EIAs (Environmental Impact Assessment) and EMPs (Environmental Management Plans).

The risk assessment process is becoming more common in industry because of the use of ERA in regulation and management practises. A risk management plan is usually developed after a detailed risk assessment process, to evaluate alternative risk reduction and prevention measures and to implement cost-effective options. ERA can assist managers in tasks such as compliance with legislation, financial planning, site-specific decision-making, prioritisation and evaluation of risk reduction measures, and precautionary or remediation actions (DEAT, 2000).

There is a substantial overlap in the ERA and EIA process, meaning that the ERA framework can be integrated with the generic EIA procedure. Examples of overlap and the use of ERA within an EIA process are as follows (DEAT, 2000):

- Hazard identification takes place within an EIA. If significant uncertainties are identified, the EIA may need to be extended to include an ERA.
- ERA can be used to determine the magnitude, extent, severity, uncertainty and variability of significant impacts identified by the EIA.

ERA applications that can be used within a risk framework therefore include the following (DEAT, 2000):

- Site-specific decisions.
- Comparative risk analysis and determining alternative risk options.
- Determining acceptable risks to develop environmental standards or benchmarks.

The value of ERA within the EIA structure can therefore easily be validated and its usefulness demonstrated. Areas that require clarification and support are the role of stakeholder involvement, and the interactive approach between government, interest groups and public participation (DEAT, 2000). The establishment of government institutions or divisions to support the integrated use of EIA and ERA within an IEM framework are critical.

1.3.2 Ecological Water Requirements: Resource Quality Objectives and ecospecs

The current approach by which Ecological Water Requirement (EWR) studies are conducted are conceptually risk-based, but the practical application of a risk-based methodology still has to be formulated and implemented.

Resource Quality Objectives (RQOs) are set for components of the resource during an EWR study and relate to certain risk levels. Where resources need a high level of protection, a strict set of objectives that will represent a low risk of damage to the system, will be set. Adopting a risk-based approached to EWR therefore provides a nationally uniform basis for deciding on the acceptability of impacts, while at the same time allowing natural site-specific differences to be taken into account by setting resource-specific objectives (Jooste *et al.*, 2000).

Ecospecs (ecological specifications) are clear and measurable specifications of ecological attributes that define the Ecological Reserve Category (ERC) and serve as an input to RQO. Ecospecs refer explicitly and only to ecological information, whereas RQOs also include economic and social objectives (IWR Environmental, 2003).

The critical components of the RQOs are:

- Requirements for water quantity, stated as flow requirements for a river reach or estuary, and/or water level requirements for standing water or groundwater, and/or requirements for groundwater level in order to maintain spring flow and base flow in rivers and other ecological features.
- Requirements for water quality (chemical, physical, and biological characteristics of the water).
- Requirements for habitat integrity, which encompass the physical structure of instream and riparian habitats, as well as the vegetation aspects.
- Requirements for biotic integrity that reflect the health, community structure and distribution of aquatic biota.

The specialist workshop conducted during this investigation (i.e. Phase II) therefore focussed on the physical drivers as components of the RQOs, i.e. water quantity, quality and habitat integrity.

1.3.3 General

For effective management, determining the risk associated with a specific hazard is vital, as the manager's response to the hazard must depend on the risk it poses (Claassen and Wade, 2001). It is therefore essential to develop an understanding of risk management and risk procedures among water resource managers, with risk management being the action when a decision is based on knowledge of the likelihood of events and their consequences.

CHAPTER 2

AIMS AND OBJECTIVES OF THE RISK-BASED OBJECTIVES STUDY

The aims and objectives of both Phases I (Jooste *et al.*, 2000) and II (current study) of the risk-based objectives (RBO) study are presented, with the links between the studies elucidated.

2.1 PHASE 1: 1998-1999

Specific aims of the Jooste *et al.* (2000) study, i.e. Phase I, were to:

- review and consolidate research into setting objectives for water quantity, water quality, habitat integrity and biotic integrity requirements of water resources, as these relate to the designation of the Ecological Reserve;
- investigate new and emerging trends in using risk concepts for setting environmental objectives;
- identify possible approaches to incorporate concepts of risk into setting integrated Resource Quality Objectives for protecting water resources; and
- identify research direction(s) addressing the development of methodologies for setting integrated objectives for water resource protection, in order to provide a key component of the current DWAF project to develop and implement a national protection-based classification system for water resources in South Africa.

Although the process for classifying water resources is currently under review, with two classification systems currently available (categories A-F, and descriptive categories such as *natural*, *good*, *fair* and *poor* (see Appendix 1 for the relationship between the two systems) (Scherman *et al.*, 2003)), the use of a risk-based approach for classification is applied in both classification systems. The following factors are inherent to the classification project and set the framework for the initiation of the risk-based objectives project (Jooste *et al.*, 2000):

- Not all water resources have the same level of protection, with each Ecological Reserve Category (ERC) and Ecological Management Class (EMC) carrying specific levels of protection or levels of *risk* of damage to the sustainability of the ecosystem.
- Resource Quality Objectives will be set for each water resource. RQOs are a statement (numerical or descriptive) of requirements for a given level of protection, and will be set for water quantity, water quality, habitat integrity and aquatic biota, as they relate to the designation of the Ecological Reserve.

The report for Phase I (Jooste *et al.*, 2000) was presented in three parts, with Part 1 comprising background literature on risk concepts and the feasibility of using of a risk-based approach for water resource management. The integration of risk objectives and risk criteria (acceptable risk), with ecological and management objectives, was discussed. Part 1 concluded with identifying research needs for the effective use of a risk-based approach to set integrated environmental objectives. Subjects covered included the need for risk management structures and policy, and the importance of understanding risk concepts and improving its accessibility to practitioners and managers. The integration of co-occurring stressors was identified as an important research area. The value and importance of collecting fundamental southern African biological and ecological data, in order to improve our understanding of ecosystems and stressor-response relationships, was also emphasized.

Parts 2 and 3 of Jooste *et al.* (2000) summarised the findings of two specialist workshops. For the first workshop (Part 2) literature was reviewed for information on functional relationships that exist between selected stressors and biotic response, i.e. can the occurrence of a stressor be related to an observable biotic effect. The following variables were selected for review:

- water quantity (flow)
- water quality, in the form of:
  - toxics
  - nutrients - nitrate, nitrite, ammonia, phosphate, iron, manganese
  - system variables - pH, electrical conductivity, salinity, Total Dissolved Solids (TDS), Total Suspended Solids (TSS), temperature
- habitat

Part 3 documented the discussion and findings of a workshop on using risk-based objectives to set flow requirements for rivers. The selection of flow as the first parameter under discussion was due to the Instream Flow Requirement (IFR) team already implicitly following a risk-based approach when determining water quantity requirements during flow estimates and Ecological Water Requirements studies. Documents written by Hughes and O'Keeffe were included as a first attempt at developing a framework for determining the water quantity Reserve, and defining different levels of flow-related stress for instream riverine fauna. The stressor-response method explored in this section has been finalized and published (O'Keeffe *et al.*, 2002).

2.2 PHASE II: 2000-2001

One of the key features of a risk assessment, particularly in characterizing and evaluating the probability of effect, is information on *stressor-response relationships (SRR)*. Conventionally, the effect of a stressor is measured in a controlled laboratory environment using a single species or few selected species, i.e. toxicological information. As this information is then extrapolated from one species in the laboratory to the same species in the river, and to many other species, populations, communities and the ecosystem, it is important that stressor-response information be available, and relationships be characterized and quantified. SRR therefore need to include the involvement of the end-point, and the direction of change in both stressor and response. As SRR were developed for flow during Phase I of the RBO study, water quality and habitat were the focus of Phase II, although flow relationships were also discussed.

The primary focus of Phase II of the RBO project was therefore a workshop held during February 2000 with the overall aim of *linking stressors and associated responses to an identified end-point, and integrating stressor responses for co-occurring stressors.* The stressors considered were changes to *flow (water quantity)*, *water quality* and *habitat.*

The following *discussion points* were identified and defined to lead the workshop:

- If the aim is to integrate stressor effects, a common end-point (e.g. the sustainability requirements of the Ecological Reserve) must be defined.

- How should stressors be defined so that a given response can be linked to the stressor stimulus, and can the response be related in some way to the identified end-point?
- What techniques, procedures or protocols already exist to determine stressor exposure and effects?
- Can risk-based objectives (quantitative or semi-quantitative) reasonably be derived for use within the present regulatory framework?

Outcomes of discussions surrounding these identified questions are detailed in Sections 3.2 to 3.5.

CHAPTER 3

PHASE II: RISK-BASED OBJECTIVES WORKSHOP

3.1 INTRODUCTION

South Africa has traditionally dealt with stressors on an individual basis, e.g. setting water quality guidelines for toxic chemicals. Although valuable, this approach has shortcomings as integrated, cumulative, antagonistic or synergistic effects of individual chemicals (for example) are not managed or monitored. Effect-specific criteria, and not only substance-specific criteria, therefore need to be developed. By implementing water resource strategies that focus on impacts on the resource, e.g. resource directed measures, a risk-based approach is implemented and resource-based management objectives are set. However, for the management of co-occurring stressors, it is important to define stressor-response relationships for individual and co-occurring stressors. Before these relationships can be evaluated, a number of fundamental questions need to be answered. Workshop results are reported as a report-back per identified question.

Jooste and Claassen (2000) produced a discussion document to lead the workshop. To focus thinking, the *minimum requirements for undertaking risk assessments*, were identified.

- The assessment should support the management decisions to be made. This alignment should be explicit in the risk assessment planning phase and be evident in the assessment results.
- The end-point(s) selected for the assessment should be appropriate in terms of both its ecological importance and its relevance to the management objectives.
- A risk hypothesis should be formulated that includes the stressor source and exposure routes, exposure-effect relationships, ecological end-points and ecosystem processes.
- An analysis plan should be drawn up and reviewed in terms of its alignment with management needs and scientific rigour.
- A critical evaluation of all available information should be conducted to assess weaknesses and identify areas where more information needs to be collected. Variability and uncertainties should be explicitly assessed and accounted for throughout the assessment.
- The characterisation of exposure should include descriptions of the pathways, spatial and temporal attributes and account for variability and uncertainties.
- The characterisation of effects should establish causality between exposure and effects and have clear links to the end-points.
- The estimation of risk should integrate the exposure and effects information and provide a sound basis for evaluating the hypothesis. Different lines of evidence should be evaluated to support the conclusions.
- Communication of the results should be clear and provide sufficient information on all the phases of the assessment to allow for peer review.
- Scientific rigour should be upheld throughout the assessment.

3.2 QUESTION 1: DEFINE A COMMON END-POINT

During a risk study, it is essential to define and describe the ecosystem or factors at risk according to structural or functional relationships. Co-occurring stressors, e.g. flow, toxics, habitat, must therefore all be assessed to the same common end-point in order for risk-based objectives to be applied. In general, end-points should (Jooste and Claassen, 2000):

- reflect important ecological characteristics of the system;
- be susceptible to known or potential stressors; and
- be relevant to management goals.

Once a common end-point has been determined for co-occurring stressors, i.e. the entity considered to be of ecological value, potential assessment end-points need to be identified, i.e. characteristics of the entity that are at risk. *Assessment end-points are therefore the definitive measures that scientifically and ecologically represent broader management concerns* (Claassen *et al.*, 2001). The next step would be to determine measurement end-points, e.g. specific values generated by field or laboratory tests, such as LC50s (i.e. the lethal concentration responsible for the mortality of 50% of a population).

For the purpose of discussion at the workshop, the common end-point was identified as the sustainability requirements of the Ecological Reserve (see Section 1.3 of Jooste *et al.* (2000) for a discussion around concepts of sustainability and resilience). As each Reserve category has a characteristic profile of ecological integrity and set of objectives, it is necessary to define exactly what is meant by the *sustainability requirements of the Ecological Reserve.* Debate ensued around the meaning and application of this concept, e.g. is it the risk of moving from an assigned category to a lower category, and therefore not maintaining the ecological function associated with the assigned category; is it the risk of impacting on the resource base (that is, that level below which recovery will not be possible); is it the risk of not meeting RQOs per category; or is it the risk associated with the uncertainty of defining the Ecological Management Class (EMC). It must be emphasized that these are management risks rather than ecological risks, although the EMC is derived from ecological considerations.

The final conclusion reached by the workshop was that as Ecological Reserve Categories are defined as deviation from the natural state (or the natural stress profile), the risk would be that of changing category or losing ecological function, and the ecological end-point is to maintain conditions as required by the assigned category (therefore incorporating the *probability* of changing to a lower category).

3.3 QUESTION 2: HOW SHOULD STRESSORS BE DEFINED SO THAT A GIVEN RESPONSE CAN BE LINKED TO THE STRESSOR STIMULUS, AND CAN THE RESPONSE BE RELATED IN SOME WAY TO THE IDENTIFIED END-POINT

A hazard is an event (e.g. concentration of a chemical) that is detrimental, while stressors also have natural stress profiles. The hazard therefore causes a degree of deviation from the natural stress profile (or reference condition). The degree of deviation is considered as the end-point or *effect*, which is measured as the *response* (e.g. change in abundance, life-stage or survival). Responses should be recorded at the highest available level of organisation – extrapolation will still be needed if this is at the level of organism.

The aim was therefore to develop a *stress index* for each variable identified by the workshop groups, and develop a *descriptor* for each stress level. This approach was developed during Phase I of the RBO project for flow, and is encapsulated in a research paper by O'Keeffe *et al.* (2002). Further development of the approach gave rise to the Flow Stressor-Response (FS-R) method used to determine low flows during EWR studies. The principles of this method are shown in Section 3.3.1. The aim of this component of the workshop was therefore to develop a similar stressor-response relationship for variables other than flow.

Comment: The use of the term *stressor* in this chapter is not in complete agreement with the definition in the glossary and international usage. In this chapter a *stressor* is understood to not always have a negative effect, and only when the magnitude and frequency of a stimulus exceeds its safe bounds does it become a stressor. However, according to international literature, a stressor is already assumed to be causing an adverse effect.

3.3.1 Water quantity: Flow Stressor-Response method for determining low flows - an overview

The excerpt regarding this method is taken from IWR Source-to-Sea (2004) and O'Keeffe *et al.* (2002).

The FS-R method was designed to guide the evaluation of the ecological consequences of modified flow regimes, based on the principles of ERA, and uses an index of flow-related stress. Some of the indicators used in assessing the levels to which a river is stressed are fish, invertebrates and vegetation (O'Keeffe *et al.*, 2002). As the impact of too little or too much flow at the wrong phase of the hydrological cycle is the stressor, the focus is on flow-dependent biota. The FS-R method recognises natural stress, as low-flow episodes are part of the natural disturbance regime of a river.

The basis of this method is the application of a generic stress index from 0-10 (see Table 3.1 for selected stress levels and associated descriptions), which describes the progressive consequences of flow reduction to the flow-dependent biota and river processes. The stressor, i.e. unnatural flow patterns at the wrong times, and resultant hydraulics and habitat changes, are related to biotic stress responses in terms of abundance, life stages and persistence / survival. These relationships are translated into a stress profile for any flow regime, in terms of magnitude, frequency and duration. Examples of the stress index development for flow-dependent biota are shown in Table 3.1.

Table 3.1 Selected levels of the generic dimensionless stress index developed by O'Keeffe *et al.* (2002).

Stress Index	Stressors		Responses		
	Flow	**Habitats**	**Abundance**	**Life-stage**	**Survival**
0	Very fast, very deep.	All very abundant.	All very abundant.	All healthy.	All species.
2	Fast, deep, but slightly reduced.	Critical habitats not abundant.	Slight reduction for rheophilic species.	All healthy in some areas.	All species.
5	Moderate / slow, few deep areas.	Critical habitat very reduced.	Remnant populations of all rheophilic species.	Critical life stages of sensitive species non-viable.	All species.
7	Slow, shallow.	No critical habitat.	All rheophilic species rare.	All life-stages of sensitive species at risk or non-viable.	Sensitive species disappear.
10	No surface water.	Sub-riverbed refugia only.	Only specialist survivors.	Virtually no development.	Only specialist survivors.

The use of the FS-R method therefore broadly consists of the following steps (IWR Source-to-Sea, 2004):

- A stress index of 0-10 is described for each element of the biota (fish, invertebrates, riparian vegetation) in relation to stresses experienced by specific flow-dependent organisms / processes under different flow conditions, and attached to responses to changing flow and habitat conditions.
- Each stress value is attached to a specific flow per site, via the hydraulic calibrations for the site.
- Natural and altered low-flow time series are converted to stress time-series, and then to stress duration curves and spell analyses. This information is used to evaluate any low-flow scenario provided.

Comment: As the method is reported in O'Keeffe *et al.* (2002), it is only informally risk-based. Further developments may include a description of what flow characteristics result in what level of impact on the target, e.g. a wet season flow of more than 20% lower than the normal flow may result in a small loss of individuals in flow-sensitive species, and 50% below normal flow may result in a significant loss in flow sensitive species. From these data and the likelihood of the occurrence of these flow conditions, the risk is calculated.

It should be noted that studies such as the Thukela EWR study have created the opportunity for a number of these developments to take place. The data therefore exists for risks to be calculated.

3.3.2 Water quality workshop session

Workshops held as part of Phase I of the RBO project highlighted the dearth of information on functional relationships between water quality stressors and responses. Problems identified during Phase I were as follows (Jooste *et al.*, 2000).

- Most toxicological data are produced using small groups of individuals from a species. Few data exist that relate to effects on a whole population, meaning that extrapolation is required.
- Most studies concentrate exclusively on sensitive or susceptible life-stages of organisms.
- Only a few end-points are well reported, e.g. the most widely used end-points of mortality and fertility for invertebrates, and biomass density for bacteria and phytoplankton.

In the water quality field most data is therefore hazard-based, and incorporates only a minimal component of frequency and duration. The focus of the water quality component of the workshop was therefore to attempt to develop a stress index and investigate stressor-response relationships for selected variables, following the method used by the flow team, i.e. Section 3.3.1. The degree of deviation from the *reference condition* or *natural stress profile* (possibly more accurately called a *natural stimulus profile*, according to internally accepted definitions of a stress (see comment in Section 3.3)) (which will not be an instantaneous measurement, but probably a median of the relevant data record) was acknowledged to be the end-point or effect, with the (anthropogenic) hazard causing the degree of deviation and the end-point measured as responses to the hazard. The aim was to specify *instantaneous levels* of hazards (normally at a particular flow) for the following stressors (see list below), and to integrate rate of change and duration at a later stage. It was also recognized that specific descriptors need to be developed for each stress level.

- toxics
- system variables
- nutrients

Information was generated for selected stress levels only i.e. 0, 2, 5, 7 and 10 – those levels where greatest changes are thought to take place. The same responses are assumed as shown in Table 3.1. Surrogates (e.g. algal responses for nutrient index) and best available knowledge was used in developing the stress tables, which affects the uncertainty of the result.

Note: The system-specific calibration of these indices is essential, as the stress index is meaningless without calibration. The information reflected here was based on a single workshop event, and data have therefore not been tested or refined.

<u>Stress Index: Toxics</u>

Table 3.2 represents a first attempt at developing a stress index for toxics.

Table 3.2 Stress index levels and qualitative descriptions of the associated toxic levels.

Stress Index	Qualitative description of the stressor / hazard
0	Level of toxic is the same as natural / reference condition.
2	Negligible level of toxic as compared to natural / reference condition.
5	Low level of toxic as compared to natural / reference condition.
7	Moderate level of toxic as compared to natural / reference condition.
10	High level of toxic as compared to natural / reference condition.

Note: *Negligible, low, moderate* and *high* must be calibrated and will be system-specific.

Stress Index: Nutrients

Table 3.3 represents a first attempt at developing a stress index for nutrients. Algal responses are used as a surrogate for nutrient responses.

Table 3.3 Stress index levels and qualitative descriptions of the associated nutrient levels.

Stress Index	Qualitative description of the stressor / hazard
0	Characteristic natural levels that represent algal productivity in terms of magnitude and ratio.
2	Nutrient levels that do not cause a change in primary producers, i.e. nutrient LOEC (lowest observed effect concentration).
5	Nutrient levels causing a change in heterotrophs.
7	Nutrient levels causing a change in habitat and heterotrophs.
10	Nutrient levels causing a system change.

Stress Index: System variables

Table 3.4 represents a first attempt at developing a stress index for system variables, e.g. pH, electrical conductivity, dissolved oxygen.

Table 3.4 Stress index levels and qualitative descriptions of the associated levels of system variables.

Stress Index	Qualitative description of the stressor / hazard
0	Level of system variables is the same as natural / reference condition.
2	Negligible deviation from natural / reference condition.
5	Low deviation from natural / reference condition.
7	Moderate deviation from natural / reference condition.
10	High deviation from natural / reference condition.

Note: *Negligible, low, moderate* and *high* must be calibrated and will be system-specific. Populating these values (Tables 3.3 and 3.4) will follow a different approach to toxics as they represent ranges, and responses to changing levels of nutrients and system variables are U-shaped (U-shaped responses may have to be split as, for example, high or low nutrients will have very different impacts). A number of databases exist with toxics data, but very little work has been conducted on determining functional relationships for system variables or nutrients. This need was identified as **critical**.

Approach for conducting a water quality assessment utilizing the concepts of stress and risk

The following preliminary water quality approach was developed – see Figure 3.1:

Use the generated Stress Index and calibrate on a system-specific basis. Produce system-specific hazard levels.

↓

Apply to the system to produce a reference condition stress profile.

↓

Produce a reference condition duration curve.

↓

Assess altered stressor levels to derive the altered stressor time-series (using the flow-concentration relationship for the variable under evaluation).

↓

Produce the altered Stress Index time-series.

↓

Produce the altered Stress Index duration curve.

↓

Compare this information with the reference condition duration curve to assess the degree of deviation. This difference is the *hazard*.

Note: To calculate risk, set conditions for duration and the probability of exceedence.

Figure 3.1 A preliminary approach for assessing water quality using stressor-response functional relationships.

The following **knowledge gaps** were identified during the water quality process:

- A link is needed between primary and secondary producers.
- Reference conditions for water quality must be specified.
- Need to relate the degree of response to the degree of change.
- Need to define what a degree of deviation means in ecological terms.
- Developing flow-concentration relationships and models (however, the method developed by Malan and Day (2002a, b; Malan *et al.*, 2003) is available for use).
- Need to bridge the fundamental gap between water quality and quantity, i.e. water quality uses 0 stress as reference condition, while quantity does not use reference condition in its definitions.
- Water quality is represented by three separate stress indices, and not an integrated water quality index.
- Experimental research and an assessment of international databases (particularly for toxics) are needed for calibrating qualifiers of the stress index table.

3.3.3 Water quantity and habitat workshop sessions

The aim of this session was to produce stress-flow relationships for a number of components, i.e. fish, macroinvertebrates, riparian vegetation and geomorphology (functions such as channel maintenance and sediment transport), and evaluate the influences of *high and low flows* on each component. A generic rule table(s) (i.e. define stress levels 0-10 per component) therefore needed to be produced which incorporates site-specific hydraulic characters and produces stress-flow relationships. Due to the different impacts of high and low flows, it might be necessary to separate out the flows and produce generic tables for each. In some instances, seasonal stress relationships will also need to be identified and incorporated.

It was agreed that the derivation of these generic stress tables would incorporate the habitat component as well, as the interaction between flow and geomorphology (i.e. channel shape and structure) defines the habitat in which in-stream biota occur.

Examples of descriptors that could be used to populate stress tables for the different components, were as follows:

- Fish: migration and breeding rate
- Riparian vegetation: growth and recruitment
- Geomorphology: channel maintenance and sediment transport.

As for water quality, the quantity team also identified the need to interpret deviations from the stress profile in ecological terms. It was felt, however, that defining deviations in ecological terms was already incorporated in the thinking of the IFR / BBM (Building Block Methodology) approach, as well as the DRIFT (Downstream Response to Imposed Flow Transformations) approach.

The following **knowledge gaps** were identified by the water quantity (flow) and habitat groups:

- Ecological interpretation needs to be included, and further developments need to define relationships in terms of ecological end-point.
- Physical impacts on habitats, e.g. bulldozing, need to be addressed in some way. It may be possible to use available data to generate stressor-response metrics.
- An integrative methodology may be needed for the DRIFT and BBM approaches.

It is important to note that a number of the points mentioned in this section are currently being addressed by existing projects, including refinement and development during Ecological Water Requirements studies, e.g. the Thukela study initiated in 2001.

3.4 QUESTION 3: WHAT TECHNIQUES, PROCEDURES OR PROTOCOLS ALREADY EXIST TO DETERMINE STRESSOR EXPOSURE AND EFFECTS

These requirements have largely been identified as knowledge gaps, and are being covered by a range of projects currently being conducted or completed. See Section 1.2 for a number of tools available internationally.

A focus for all projects is to identify tools, methods, research and information in relation to ecological functioning, stressors and ecological effects.

3.5 QUESTION 4: CAN RISK-BASED OBJECTIVES (QUANTITATIVE OR SEMI-QUANTITATIVE) REASONABLY BE DERIVED FOR USE WITHIN THE PRESENT REGULATORY FRAMEWORK

With the river classification system (and associated resource-directed measures for management) being conceptually risk-based, a regulatory environment has been established that is accommodating of risk-based assessment and management. The use of source-directed controls could however benefit from a risk assessment approach, particularly in developing scenario-based decision support systems (Jooste and Claassen, 2000). The risk-based approach is therefore essentially an extension of the precautionary approach as adopted by DWAF.

Although risk-based objectives can be used in the following process;

- identify hazard,
- quantify the hazard, and
- express the likelihood of effect on the identified end-point,

the question remains as to its application and usefulness. It was felt that the usefulness of this approach is largely dependent on current management approaches and thinking (although current thinking is in line with current approaches, e.g. to EWR assessments). Risk communication and the development of a risk policy, were identified as essential management requirements.

CHAPTER 4

RECOMMENDATIONS AND IDENTIFIED GAPS

- More focus is required on the definition of stressors. Some fundamental research on the water quantity and quality conditions that arise naturally without negatively impacting on biota, as well as those conditions which do affect biota, needs to be identified. Since this would likely involve observations in real ecosystems, it poses a real challenge to the research- and funding communities, but one that would be well worth its investment in improving confidence in resource management.

- Develop ecosystem knowledge and understand the behaviour of ecosystems and their response to stressors. This knowledge arises from monitoring, experimentation or modelling (among other methods), and must incorporate the impact of the environment on the behaviour of the stressor. A good knowledge of the system quantifies variability, increases confidence in results, and improves the predictability of the assessment.

- Cause-effect diagrams facilitate understanding of the study. Factors such as sources, stressors, exposure routes, end-point, response, measure and ecosystem links must therefore be defined (Claassen *et al.*, 2001).

  Examples modified from Claassen *et al.* (2001) include the following:

Sources	Effluent discharge
Stressors	Heavy metals
Exposure routes	Speciation / transport
End-point	Aquatic invertebrates
Response	Mortality
Measure	Presence – absence
Ecosystem links	Fish

- Introducing hazard assessment to an extensive site / situation-specific risk assessment approach: It is recommended that a *tiered assessment approach* be adopted. The criteria to move from tier to tier needs to be formulated, and although such approaches have been developed (e.g. Direct Estimation of Ecological Effects Potential (previously known as TEHA or Toxicity-based Ecological Hazard Assessment)), the adoption of such approaches has not yet taken place although being investigated (Jooste and Claassen, 2000).

- Most risk assessment protocols allow for participation by *Interested and Affected Parties* (I&APs). Guidelines need to be established to decide the extent and representation in the risk management process (Jooste and Claassen, 2000).

- Management decision on fundamental *assessment hypothesis:* The statistical foundations of risk assessment allows for different hypotheses in assessing stressor impacts. On the one hand, it can be assumed that an effect or a risk exists until the contrary is proven, or conversely, it can be assumed that no effect or risk exists until the contrary is proven. This would need to be reflected in management policy (Jooste and Claassen, 2000).

- Setting of *bright lines* in the risk continuum to denote points or levels corresponding to, for example, clearly trivial (*de minimis*) risk and clearly unacceptable (*de manifestos*) risk. This would serve to divide the risk continuum into action domains, and will largely be a matter of policy, possibly in consultation with I&APs (Jooste and Claassen, 2000). There exists a considerable body of knowledge in both the USA and Europe on the psychology of risk criteria and how these could be developed, but these need to be adapted to South African conditions.

- *Peer review* is an objective of the risk assessment process, and is relevant particularly in the case of disputes (Jooste and Claassen, 2000).

- A gap that was identified during the workshop was the need for a *pilot application using actual generated data*, i.e. calibrating hazard or stressor-response relationships using real data. This gap therefore concerns a need to validate methods so as to set stressor-response functions for all variables

- The impact of *biological stressors* have not been evaluated, e.g. exotic species, introduced species (e.g. via water transfers), and genetically modified organisms. These impacts are of particular importance in a management framework.

- How will *mixtures* be dealt with? It may be possible to develop a generic stress index for some key effluents, e.g. textile and pulp and paper effluents; even on a site-specific basis. Care should be taken in calibrating such a stress index, due to the variability of effluents.

- The link between risk assessments and other resource-based approaches such as river health and ecosystem integrity will have to be defined, so as to assess the validity of a risk assessment approach.

- A large gap in thinking is still the development of *co-integrating the effects of stressors towards defining ecological risk*. More research needs to be undertaken to distinguish mechanistic issues from expectation issues, as risk is essentially an expectation issue. Methods exist in which expectations can be integrated (Jooste, 2001), but these need to be validated. However, the different stressors (each of an innumerable number of water quality-related stressors, flow, a number of habitat-related stressors, as well as a number of biotic stressors) may share mechanistic ecosystem pathways. This would mean that mechanistic phenomena such as additivity, supra-additivity (synergism) and infra-additivity (antagonism) might be involved. These issues should be investigated so that SRR for not only each individual stressor can be derived, but also changes to the SRR in the presence of other stressors.

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APPENDIX 1

The table below shows the relationship between the alternative descriptions for river health categories (Scherman *et al.*, 2003).

Ecological Condition Categories	**Ecological Management Classes**
A	Natural
B	Good (A/B, B, B/C)
C D	Fair (C, C/D, D)
E F	Poor

TRACKING MOVEMENT OF LARGE FISH SPECIES THROUGH A RIVER SYSTEM: METHODS DEVELOPMENT

DRAFT REPORT





Prepared for:
WATER RESEARCH COMMISSION (WRC)
South Africa

B.R. PAXTON
UNIVERSITY OF CAPE TOWN

Water Research Commission
Pretoria
South Africa

University of Cape Town
Cape Town
South Africa

TRACKING MOVEMENT OF LARGE FISH SPECIES THROUGH A RIVER SYSTEM: METHODS DEVELOPMENT

(DRAFT REPORT APRIL 2004)

PROJECT NO. K8/536

By

BRUCE R. PAXTON

Freshwater Research Unit, University of Cape Town

Prepared for:

WATER RESEARCH COMMISSION

EXECUTIVE SUMMARY

Planned water-resource developments on the Doring River, a major tributary of the Olifants River, Western Cape, have raised concerns that declines in populations of threatened endemic freshwater fish may be accelerated. By reducing the amount of running water habitat through regulation or inundation, or preventing fish from reaching critical habitat units such as spawning or nursery areas, dams may compromise the ability of fish populations to grow, survive and reproduce. Of the latter two impacts, the Department of Water Affairs and Forestry (DWAF) has identified barrier effects of proposed dams on the Doring River as the most serious concern. Anecdotal evidence suggests that, during the earlier half of the 20th century, endemic fish populations formed large breeding aggregations in the Olifants River. No information exists on the extent to which these populations depended on extensive migrations between the two rivers, or up and down either, or what advantage migration may confer in terms of reproductive success or recruitment. In order to attempt to address this uncertainty, a tagging study has been underway in the catchment since 2001 to determine the extent of movement between reaches in the Doring River. This has been largely unsuccessful due to the low numbers of fish in the system and the large amount of effort required to produce any results.

A literature review for the Water Research Commission on methods for studying the spatial behaviour of fish (Paxton 2004) identified telemetry as the most effective way of acquiring information on movement and habitat use by adult fish at spatial and temporal resolutions that would address management concerns. A joint study is being planned by the Freshwater Research Unit (FRU) at the University of Cape Town, the Norwegian Institute for Nature Research (NINA) and the South African Institute for Aquatic Biodiversity (SAIAB) to track the Clanwilliam yellowfish *Labeobarbus capensis*, sawfin *Barbus serra* and Clanwilliam sandfish *Labeo seeberi* in the Doring River, by means of radio telemetry. Before tracking can commence, however, the response of the target species to capture, transport and surgical implantation of radio telemetry transmitters needed to be ascertained and the logistics of the tracking procedure needed to be planned based on a detailed knowledge of the study area. Non-lethal methods of capture, marking and acquiring biological information from the fish also need to be developed. Ideally the reaches where the telemetry is proposed to take place need to be mapped and physical habitat for the indigenous fish described.

In terms of the agreement between the Water Research Commission and the University of Cape Town, (K8-536), the primary aim of this study was to lay the groundwork for telemetry studies to be conducted on the threatened endemic fish species of the Olifants and Doring Rivers. To achieve this aim, the following objectives needed to be met:

(1) establish the effects of (a) capture, (b) tagging, and (c) transmitter implantation, on the study species: the Clanwilliam yellowfish *Labeobarbus capensis*, sawfin *Barbus serra* and Clanwilliam sandfish *Labeo seeberi*;
(2) develop methods for acquiring biological information (particularly their sex on the basis of external morphology) on tagged fish using non-lethal methods;
(3) describe the physical conditions of the study area;
(4) access funds for radio telemetry studies.

In order to address Objective 1a above, i.e. to develop methods for capturing the native fish, fyke nets were evaluated as an alternative to gill nets. While they were successful in that they limited injury to the fish compared with gill nets, catch rates were much lower. Catch rates of non-native species (bluegill sunfish and smallmouth bass), however, tended to be higher, suggesting that fyke nets are selective for these species.

To address Objective 1b, VI Alpha tags were evaluated as an alternative to the T-bar anchor tags used in earlier surveys for marking captured fish. Because of their small size and insertion beneath the skin of the fish, their impacts on fish behaviour and survival were considered to be far less than T-bar anchor tags. , The difficulty of inserting the tag however, together with the longer processing time, limits the number of fish which can be marked. This may also limit their usefulness for widespread application by non-technical personnel such as recreational anglers which would be important for any long-term tagging programme to succeed.

To address Objective 1c, trial runs on captive fish using dummy radio-telemetry transmitters were undertaken at the University of Cape Town and the Two Oceans Aquarium in collaboration with the NINA. The trials suggested that the target species (Clanwilliam yellowfish, sawfin and sandfish) would recover from surgery and insertion of transmitters should telemetry studies take place.

There was insufficient time to map the lower Doring River (Objective 2) during the course of tagging studies, and insufficient numbers of fish were caught to determine whether they could be sexed on the basis of external morphology (Objective 3). The search for funds for the radio telemetry study is ongoing (Objective 4).

Funds from a related project on the Doring River (Western Cape Olifants/Doring River Irrigation Study, WODRIS, PGWC 2004), to assess the likely impacts of water-resource development on fish populations in the lower reaches of the river, enabled the purchase of the specialised equipment and for more extensive fieldwork to take place. The results of that survey, and recommendations made to the Provincial Government of the Western Cape (PGWC), have been incorporated into this report.

To meet the objectives of the WODRIS study, which was to determine whether there was any movement by individual fish between the Olifants and Doring Rivers, a tagging programme was carried out between May and December 2003. Although the tagging programme yielded no recaptures, the surveys provided a more detailed picture of fish species distribution in this region and also the opportunity to experiment with the new capture and marking techniques. In the absence of information on movement, however, recommendations to the DWAF regarding the impacts of dams on the lower Doring River have been made on the basis of best available knowledge and literature reviews of related species.

These recommendations are:

Dam at Melkboom:	Unlikely to represent a major barrier to fish movement at the current levels of fish in the lower Olifants River.
Dam at Melkbosrug:	Significant populations of yellowfish, sandfish and unusually large numbers of adult sawfin persist in the middle and lower Doring River. A dam located here would represent a barrier to fish movement.
Abstraction weir:	As for a dam at Melkbosrug.

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1. INTRODUCTION

1.1 BACKGROUND TO THE STUDY

The Department of Agriculture of the Provincial Government of the Western Cape (PGWC) initiated the *Olifants/Doring River Basin Study* (ODRS) to investigate the most economical and environmentally sustainable options for development in the Western and Northern Cape in order to address the high levels of unemployment amongst historically disadvantaged communities living here. In 1998, the ODRS identified part of the coastal plain between the Olifants River and the Atlantic coast, the Aties Karoo, Klawer and Melkboom in the Western Cape for agricultural expansion using surface water from the lower Doring River and/or groundwater. In line with integrated catchment management objectives, a more comprehensive study, the *Western Cape Olifants/Doring River Irrigation Study* (WODRIS), followed on from the ODRS, aiming to examine water development options in more detail. This study identified the most viable sites for storing water as being along a 38 km segment of the Doring River from its confluence with the Olifants River to the Brandewyns River **Figure 1.1**.

Water storage and abstraction facilities in this region, however, are likely to compromise the survival of three threatened freshwater fish species: the Clanwilliam yellowfish *Labeobarbus capensis* (Vulnerable VU A1ce); sawfin *Barbus serra* (Endangered EN B1 +2abde, C1) and sandfish *Labeo seeberi* (Critically Endangered CR A1ace) (IUCN 2003). It is suspected that a dam would cut off the interchange of individuals between the lower Olifants and Doring Rivers, and would also ultimately reduce recruitment by preventing adults in breeding condition from reaching spawning sites on the Doring River in spring. However, despite circumstantial evidence for fish migration in these rivers (e.g. Harrison 1976), very little is known about the extent to which the indigenous fish depend on connectivity between river reaches for their growth, survival and reproduction.

In 2001, therefore, a series of fish surveys began which was aimed at providing greater insight into the status, distribution and movement of native fish populations in this river system. These surveys were funded by the Department of Water Affairs and Forestry, as well as by the Provincial Government of the Western Cape Department of Agriculture as part of the WODRIS study. The fish surveys focused primarily on the mainstem of the Doring River, for which very few data were then available. One of the main objectives of these surveys was to gain a better understanding of fish migration, and a tagging programme was started that, it was hoped, would yield information on the extent to which individual adult fish moved between river reaches. In this study, in February and October 2001, gill nets were used to capture adult fish, primarily from mainstem reaches on the Doring River, but also from three sites on the Olifants River. Captured fish were marked by means Floy® T-bar anchor tags inserted into the musculature of the fish below the dorsal fin and released. The 2001 surveys yielded very little information on fish movement due to low recapture rates. The distributional data acquired during the course of the study, however, were combined with historic records from Cape Nature Conservation, the South African Institute for Aquatic Biodiversity and the Albany museum to build up a picture of species occurrence throughout the catchment. The results (Paxton *et al.* 2002) highlighted several areas of concern. The absence of sawfin and sandfish occurrences in gill nets set on the Olifants River suggested that these species had become extinct in the mainstem of the Olifants River, and the very low catches of Clanwilliam yellowfish suggested that these were extremely rare. Adults of all three species (especially sandfish) were found in greater abundances in the less developed Doring River, but there appeared to be a complete absence of young adults and juveniles in all the study areas except beyond the most upstream

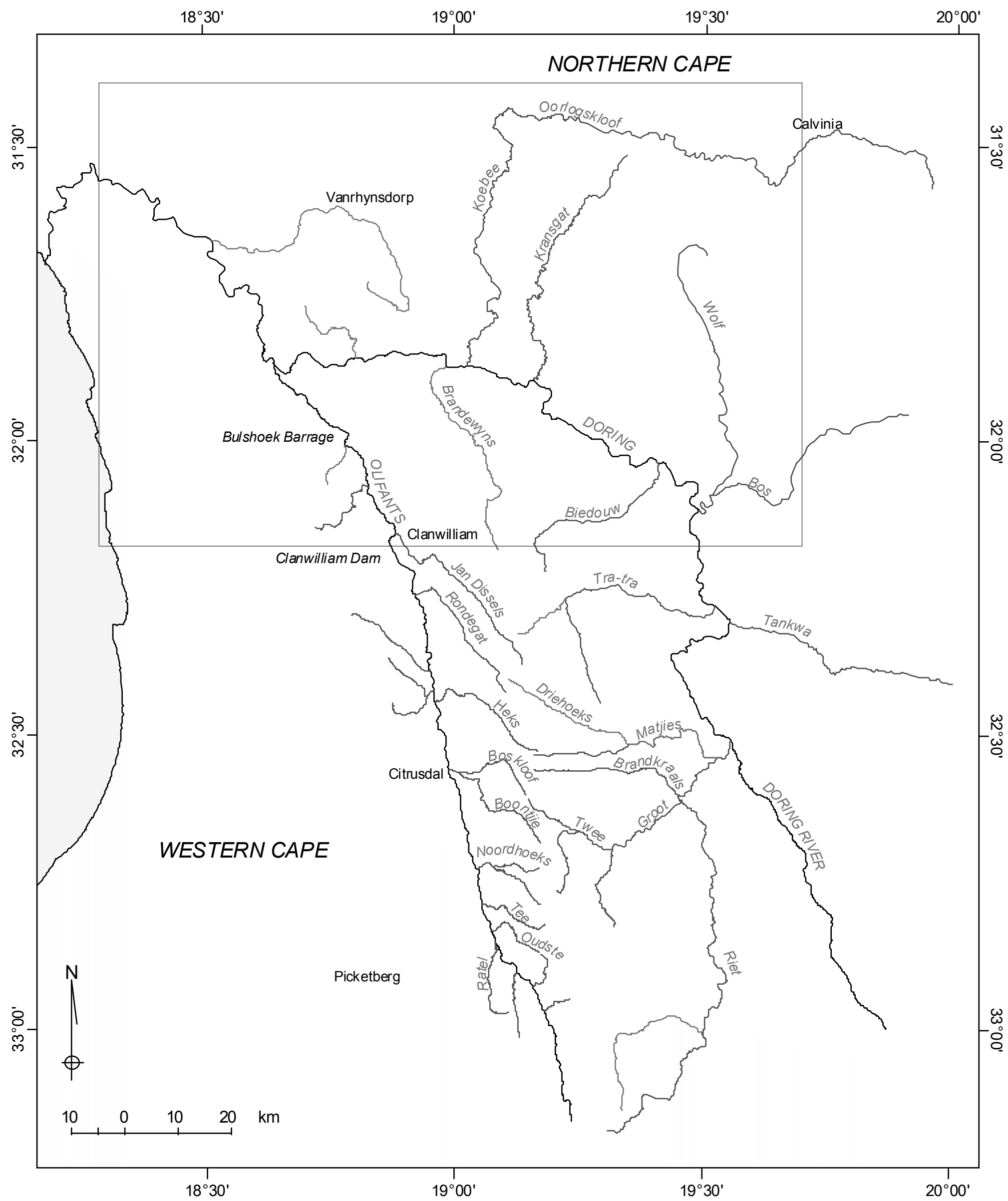

Figure 1.1 Map of the Olifants and Doring Rivers showing the major tributaries. Inset shows the study area for the current (2003) survey (**Figure 2.2**).

limit of invasion by bass *Micropterus* spp. and bluegill sunfish *Lepomis macrochirus*.

The 2002 report concluded that a more intensive, long term tagging programme spread over a number of seasons would be pre-requisite for understanding fish migration in the catchment. It also recommended experimentation with fyke nets as an alternative to gill nets, for these had caused physical damage and trauma to the fish. Additionally, T-bar anchor tags caused lesions in the skin and the resulting wounds exposed the fish to infection. As a consequence, the rate of tag loss was also expected to be high. The report recommended that alternative tagging methods would need to be investigated, and highlighted the importance of acquiring baseline ecological information on the species of concern, particularly their spawning requirements.

In 2002/2003, funds made available by the Water Research Commission, and the Freshwater Research Unit (FRU) at the University of Cape Town, made possible a second year of surveys. These surveys aimed to supplement the existing distributional database, investigate spawning areas in more detail, and investigate alternative sampling and tagging methods. Surveys were undertaken along the middle and lower Doring River, as well as in the upper Olifants River. The results from these surveys were combined with a comprehensive methods development and literature review on fish movement and habitat (Paxton 2004).

During the course of this study it became clear that because of the inaccessibility most of the Doring River, radio telemetry, combined with aerial tracking from a light aircraft, would be necessary to study fish migration and habitat use by adult fish. Early in 2002, therefore, the Norwegian Institute for Nature Research (NINA) was approached to aid design of a telemetry programme to investigate the seasonal movements of large adult Clanwilliam yellowfish, sawfin and sandfish in the lower Doring River. A proposal for such a study was developed by members of the project team (University of Cape Town), NINA and the South African Institute for Aquatic Biodiversity (SAIAB) during 2002. In anticipation of funding, however, it was agreed that it would be necessary to test the response of the native fish to the insertion of radio-transmitters, as well as identify suitable sites for catching, tagging and tracking the rare and patchily distributed populations.

Chapter 2 in this report, therefore, introduces the study sites and sampling programme of the 2003 fish surveys. Chapter 3 discusses the development and design of fyke nets for use in capturing large adult fish in the mainstem of the Doring River. Chapter 4 reports on the use of VI Alpha tags as an alternative to T-bar anchor tags for conventional tagging purposes. Chapter 5 reports on the telemetry pilot study and Chapter 6 reports on the results of the 2003 surveys. Conclusions and recommendations for further research are discussed in Chapter 7

2. STUDY SITES AND SAMPLING PROGRAMME

A comprehensive description of the Olifants and Doring Rivers catchment is provided in Paxton (2004). The inset in **Figure 1.1** (Chapter 1) highlights the 2003 study area which is shown in more detail in **Figure 2.2**. A total of 15 sites was sampled during May, September and December of 2003. Sites were distributed along approximately 90 km of the Olifants River downstream of the Bulshoek Barrage to the head of the estuary (**Figure 2.2**), and along the final 40 km of the Doring River before it joins the Olifants River near Melkboom. During May 2003, six sites were sampled on the Olifants River – two sites upstream of the Olifants-Doring confluence downstream of the Bulshoek Barrage (Klein Rietvlei *Kr* and Sandkamp *Sk*), and four sites between the Doring River confluence and the mouth of the Olifants River at: Kransgat (*Kg*), Gideonsoord (*Go*) near Klawer, and Draairivier (*Dr*) near Vredendal. The uppermost extent of tidal influence, i.e. the bridge at Lutzville, (*Lv*), was the most downstream site surveyed on the Olifants River. Three sites were sampled on the Doring River at Oudrif (*Od*), Bruinkrans, (*Bk*) and Melkboom (*Mb*) approximately 7 km upstream of the Olifants-Doring River confluence. Follow-up surveys at these same sites were planned for the months of August, September, October and December of 2003.

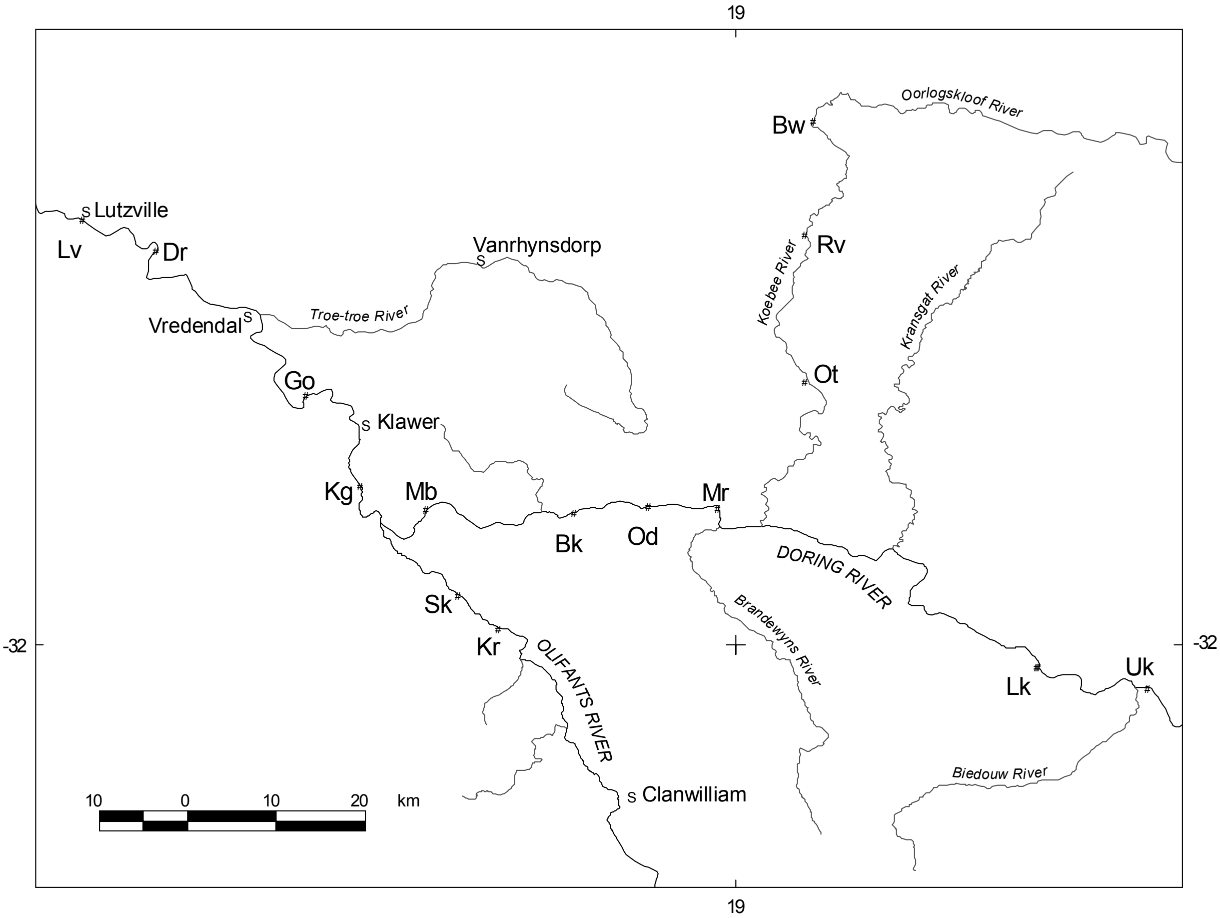

Figure 2.2 Sites sampled on the Olifants and Doring Rivers during May, September, November and December 2003.

The programme was modified after the May survey, however, after it was decided that the numbers of native cyprinids in these reaches were too low to continue with the tagging programme. Interviews with local farmers and members of angling clubs along the Olifants River during the course of the survey confirmed the findings of earlier surveys (Paxton *et al*. 2002) which showed that populations of yellowfish were extremely low and localised and that sawfin and sandfish had become locally extinct. Thus, despite suspicions that the failure of the May survey to catch indigenous fish was partially attributable to gear limitations (fyke nets were being used for the first time, see Chapter 3), the survey team felt that the study programme would need to be amended to account for the low abundances of native fish. It was decided that the number of sites needed to be reduced and effort-per-site increased, in order to focus on the lower Doring River where previous surveys had confirmed greater abundances of endemic cyprinids. All sites on the Olifants River, apart from Kransgat (*Kg*), were therefore abandoned in favour of additional sites on the Doring River. These sites included Melkbosrug (*Mr*) upstream of Oudrif (*Od*), Lankuil, (*Lk*) 42 km upstream of Melkbosrug, and Uitspanskraal (*Uk*), 14 km upstream of Lankuil. Three sites were also included on the Oorlogskloof-Koebee Rivers system because of this system's possible importance as a source of recruitment for the lower Doring River. These sites included: Ondertuin (*Ot*), the most downstream site and approximately 20 km upstream of the confluence with the Doring River, Rietvlei (*Rv*) and Brakwater (*Bw*) on the Oorlogskloof River. These sites were only visited once, during November 2003.

Sampling was not possible during August 2003 due to high flows and the first follow-up survey therefore commenced in September 2003. By the time the second survey had begun, a new net, designed and developed by members of the survey team in consultation with Australian net-makers, had been constructed. This net proved to be more effective than the previous nets and was used throughout the rest of the survey (see Chapter 3). The final follow-up survey, which included the sites listed above, was conducted during December 2003.

2.1. DATES OF THE STUDY - SUMMARY

The surveys for the study presented in this report took place between May and December 2003. The dates of the fieldtrips undertaken during the course of the study are listed below.

22-31 May 2003: Sites sampled during May 2003 with the first pair of small fyke nets included the following: Klein Rietvlei (*Kr*); Sandkamp (*Sk*); Kransgat (*Kg*); Gideonsoord (*Go*); Draairivier (*Dr*); Lutzville (*Lv*) on the Olifants River, Bruikrans (*Bk*) and Oudrif (*Od*) on the Doring River.

29 Aug-26 Sep 2003: Sites sampled from August through September with the large fyke net included: Kransgat (*Kg*), Melkboom (*Mb*), Bruinkrans (*Bk*), Oudrif (*Od*), Melkbosrug (*Mr*), Lankuil (*Lk*) and Uitspanskraal (*Uk*) on the Doring River.

03 Nov-15 Dec 2003: A second, follow-up survey of the same sites visited during September was conducted during November and December of 2003. In addition, during November, sites visited on the Koebee/Oorlogskloof system included: Ondertuin (*Od*) and Rietvlei (*Rv*) on the Koebee River, and Brakwater (*Bw*) on the Oorlogskloof River.

3. FISH CAPTURE TECHNIQUES

3.1 GILL AND FYKE NETS

Gill nets are commonly used for sampling fish in freshwater and marine environments both in South Africa and internationally. They induce high mortality rates, however, and are therefore not considered suitable for working on vulnerable or endangered populations. An suitable alternative technique for catching the large adult cyprinids in the Doring River therefore needed to be found. Different types of trap nets are available (e.g. fyke and hoop nets), the efficiency and selectivity of which differ widely from those of gill nets. Krueger *et al*. (1998), evaluating the performance of gill and fyke nets, found that fyke nets are particularly selective of cover-oriented species. Hanchin *et al.* (2002) found that fyke nets tended to select smaller fish (150 mm TL). Krueger *et al*. also pointed out that the high variability of catch per unit effort (*cpue*) among fyke net sets reduces their usefulness for detecting changes in abundance. Larger sample sizes are therefore required for statistical power. This problem may be compounded where fish abundances are extremely low.

During 2001 and 2002, gill nets were used to capture large adult fish in the Olifants and Doring Rivers (**Plate 3.1** Paxton *et al.* 2002). Four gill nets made of monofilament nylon with mesh sizes of 54, 70, 90 and 145 mm were used to sample large adult fish populations in mainstem pools. Each net was 30 m long, with a 2 m drop fitted with weighted foot ropes. The nets were set during the night after it was established that fish avoided the nets during the day. The nets were checked every hour and all indigenous fish were removed, measured, tagged and released. These nets were found to be effective for catching endemic fish in a range of size classes. The immediate and longer term effects physical damage to the fish while in the nets, however, were expected to greatly reduce their chances of survival upon release. Fish were caught in the gill nets by swimming into the net and either being wedged – held around the body; gilled – held behind the opercula; or tangled. Fish trapped in any of these ways suffered considerable trauma. Asphyxiation occurred if the net had closed around the opercula and the fish was not removed soon afterwards. At best, a loss of scales could be expected, as well as lacerations and bruising on the skin around the nape. If the fish had become entangled,

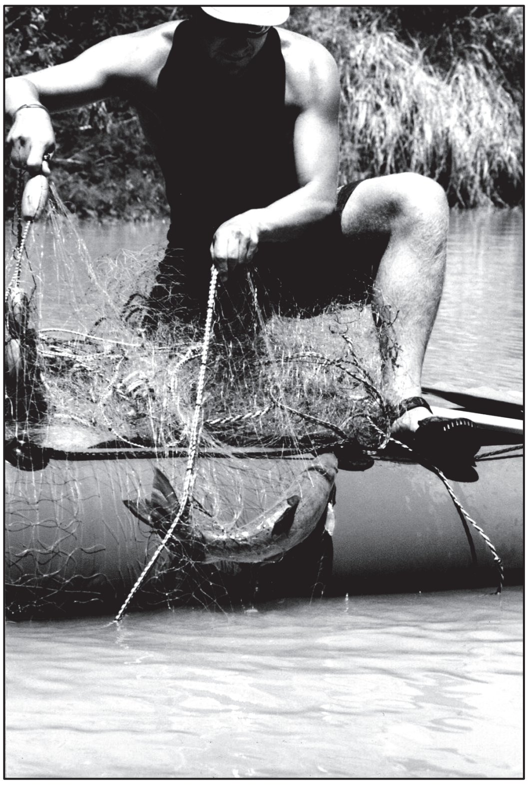

Plate 3.1 Sandfish caught in a gill net on the Doring River.

further stress could be expected from the longer handling time involved in extricating the fish.

Due to the threatened status of these fish, the use of gill-nets in this research was considered neither ethical nor effective – the success of a planned mark-recapture programme depended on the capture of large numbers of fish and their return to the river in good condition. In mark-recapture experiments, the proportion of recaptured fish is generally low (< 5%) and a high mortality of tagged individuals would further reduce the chances of recapture.

In 2002, therefore, the suitability of using other gear types to capture native species was investigated. Local and international researchers were consulted (**Table 3.1**) on a range of fish capture techniques, including trammel nets and electrofishing, before it was decided that fyke nets would be the most suitable alternative. While fyke nets have not been used extensively for research in South Africa, they have been used by fisheries research institutions elsewhere in the world to catch a wide range of different species. Their greatest advantage is that they catch fish alive and unharmed and therefore are considered ideal for working on threatened species. These nets were procured from Australia shortly before the 2003 field season began in May.

Table 3.1. Advisors consulted on fish capture techniques.

Country	Name	Organisation	Post
United States	Herke, Scott (Ph.D.)	Louisiana State University	Postdoctoral Researcher
	Loftus, William F. (Ph.D.)	United States Geological Survey Florida Center for Water and Restoration Studies Everglades National Park Field Station	Research scientist
	Nelson, Eric B.	Environmental Protection Agency (EPA) Fish & Wildlife Service Liaison	Environmental Scientist
	Sederberg, Bruce	H. Christiansen Co.	Net-maker
Australia	Ebner, Brendan	Wildlife Research and Monitoring Environment ACT	Research scientist
	Osborne, Tom	T & L Netmaking	Net maker
	Wilson, Glenn (Ph.D.)	CRC for Freshwater Ecology Murray-Darling Freshwater Research Centre - Northern Laboratory	Scientist in Charge
South Africa	Bok, A. (Ph.D.)	Anton Bok and Associates	Specialist Consultant
	Cowley, P. (Ph.D.)	South African Institute for Aquatic Biodiversity (SAIAB)	Research scientist
	Kleynhans, Neels (Ph.D.)	Institute for Water Quality Studies	Chief Specialist Scientist
	Rall, Johan (Ph.D.)	ECOSUN Environmental Consulting	Specialist Consultant
	Skelton, P. (Ph.D.)	South African Institute for Aquatic Biodiversity (SAIAB)	Director
Norway	Næsje, T. (Ph.D.)	Norwegian Institute for Nature Research (NINA)	Assistant Research Director
	Okland, F. (Ph.D.)	Norwegian Institute for Nature Research (NINA)	Researcher

3.1.1. Fyke net designs

Initially, two small two-wing fyke nets were used which were constructed from knotless knitted 6 mm green mesh with each wing being 10 m long, 1.2 m deep and fitted with a float and lead line. The fyke trap was constructed from aluminium square-framed hoops 700 × 700 mm square. The nets were anchored at three points using a combination of floats and anchor weights. The nets were set with buoys rather than stakes since there was either insufficient shallow water to push stakes in, or the bed of the river was too rocky. We used five buoys per net, one on each of the three corners of the fyke and one at the center of each wing. The net could then be held open by tying it to the bank if this was close enough. By doing this we were able to set the nets either at the surface or on the bed at depths varying between 1.5 and 3 m. The nets, set with a slight ‘V’ in the wings, were left in the water from approximately 17h00 in the afternoon till 23h00 at night, during which period they were monitored constantly. A combination of ground bait (crab, maize or flour) and light sticks were used to attract fish to the trap.

The first trial on these nets was conducted in May 2003 in the lower Olifants and Doring Rivers. Eight sites were visited between 22/05/2003 and 31/05/2003. The fyke nets were found to be effective for catching a wide range of species and sizes including: bluegill sunfish *Lepomis macrochirus*, spotted bass *Micropterus punctulatus*, two species of tilapia (*Tilapia sparrmannii* and *Oreochromis mossambicus*) and flathead mullet *Mugil cephalus*. However, the fyke nets proved to be ineffective for catching the large endemic cyprinids in the mainstem reaches which were the target of the research – all of the above species, except for the last, are exotic to the system.

During the May survey, shoals of between 20 and 30 yellowfish and/or sawfin could be observed from the banks at Oudrif (Od, Figure 2.1). On the 29th May, therefore, the two fyke nets were set end to end at the surface of the water with the trap entrances directed toward an area of the pool where fish had been observed feeding. The nets were left in the water from 17h00 till 23h00 and then collapsed overnight to avoid otter bycatch. The nets were reopened at 07h00 in the morning of 30th May and monitored till midday. After further observation, it become evident that the fish were most vulnerable to capture in a shallow region of the pool between two feeding areas between which they had been moving. The location of the fish were then established and the nets set in this shallow region, effectively cutting off one side of the pool from the other. The nets were set end to end on the bed of the river at a depth of approximately 1.5 m with the trap entrances directed towards the side of the pool where the fish were located. The fish were then herded towards the net by the survey team. An observer on the bank was able to monitor the movement of the fish in relation to the nets. The fish were observed swimming to within approximately 5 m of the nets before turning as a shoal and circumventing the team in the water. The fish stayed away from the nets for the rest of the day. We baited the nets with crab and removed them from the water at 23h00 without having had any success.

Since this survey, these smaller nets have been used in one of the tributaries of the Olifants River (the Rondegat River) where a yellowfish of between 35 and 40 mm TL was caught, suggesting that they may be effective under certain conditions – for instance if the river is narrow enough to be spanned by the whole net. Apart from the dimensions of the river channel, the failure of the fyke nets to catch the indigenous fish was ascribed to the factors listed below.

1) At the time of sampling, the water in the Doring River was clear. The nets, constructed of a knotless green mesh with a diameter 6 mm, were very visible in clear water, even at night – the catchability of the native fish may therefore increase when water is more turbid.
2) The angle of the 'V' in the wings may not have been acute enough to encourage fish to enter the trap. Because we needed to cover as much of the width of the pools as possible, the wings were set at an obtuse angle – perhaps the nets need to be set with a more acute 'V' (± 45°) and therefore longer wings would be needed in addition to a centre leader.
3) The nets did not cover a significant proportion of the depth of the pools, which ranged between 1.5 and 4 m (the distance between float and lead line was 1.2 m) – larger fish may have been more likely to swim over the top of, or underneath, the nets if such an alternative was perceived by them as being a more likely means of escape than entering the mouth of the trap – the drop of the wings therefore needed to be deeper.
4) The native fish are larger (45 – 100 cm), more active, faster and more wary of disturbance than the non-native species. They were less inclined to enter small spaces such as the entrance to the trap – the trap entrance therefore needed to be made less visible.
5) The nets were not left in the water for long enough (they were removed at night to prevent otter bycatch). It was felt that the capture rate would have increased had the nets been left in the water for longer and the fish had grown accustomed to their presence. The nets also needed to be fitted with an exclusion device for otters to prevent their drowning.

To overcome the limitations of the fyke nets used during May, a third, larger fyke net was designed and custom-built in Australia by T&L Netmaking during June and July of 2003 (**Plate 3.2**). The fyke was constructed from 18 mm mesh netting with an entrance frame 1.2 m high and 2 m wide. Five aluminium hoops, each 1.20 m high formed the trap that was 7 m long. The three wings (left and right wings and leader) were each 20 m long by 2 m high and were rigged with a leadcore bottom and floats. The fyke was anchored in the water facing downstream by means of a line rigged from bank to bank and held afloat near the trap entrance by three buoys. The wings were held open by attaching a rope from each bank, and the centre wing (leader) was held in place by means of a weight. This net was set overnight for 15 hours (17h00 – 08h00) at each site and cleared in the morning. This net, used between September and December 2003 proved more successful for catching the larger endemic cyprinids and the efficiency and selectivity of this net compared to gill nets is therefore reported in the following section.

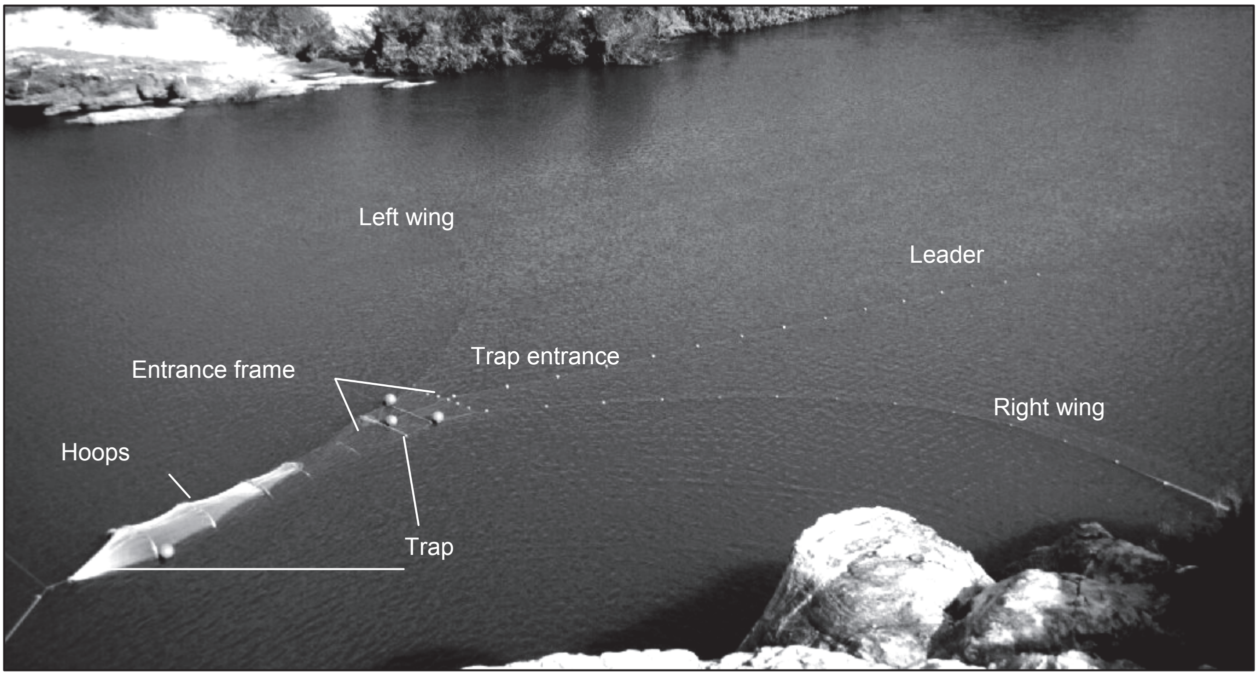

Plate 3.2 Fyke net used during the September – December 2003 field surveys. Buoys were used to keep the fyke net afloat in deep pools.

3.2 GEAR SELECTIVITY

Figure 3.1 compares mean *cpue* values (for all sites and sampling events combined) for *L. capensis*, *B. serra*, *L. seeberi*, *M. dolomieu* and *L. macrochirus* caught in gill nets (February and October 2001) and fyke nets (October-December 2003). High gill net *cpue* values during February 2001 were a consequence of the fish being confined to isolated pools over the summer months. The reduced water volume of the pools increased the density of fish and therefore catches were much higher over this period. Once the river started flowing during the winter, the fish redistributed through the system and catch rates declined. The early summer fieldtrips undertaken during October 2001 and October-December 2003 therefore reflect comparative gill and fyke net selectivity more accurately because on these occasions the river was flowing.

Despite the scaling up of the fyke net size and modifications to the design resulting in increased absolute effectiveness of the fyke nets, *cpue* values indicate that catch rates for all three of the endemic species were considerably lower than gill net catch rates, whereas catch rates for the non-native species were higher. This is probably a consequence of behavioural differences between the species – bass and bluegill are perhaps more likely to be found in proximity to cover and therefore more likely to enter confined spaces.

The high bluegill sunfish *cpue* values for October to December 2003 were partly due to the fact that sampling in 2003 extended further into the summer than did the 2001 sampling, and that there was greater fish activity during the later months (November and December). This may also be true for smallmouth bass. It should be noted, however, that these figures represent adult fish only (*M. dolomieu* >200 mm TL and *L. macrochirus* >150 mm TL) and therefore were not a consequence of increasing numbers of 0+ recruits in the summer months. A large proportion in the difference between gill and fyke net catches compared by Krueger *et al*. (1998) could be attributed to the fact that the fyke nets they used were set on the bottom, whereas gill nets were set on the surface of the water. In this study, the fyke nets were held on the surface of the water by means of buoys and were therefore set at a similar position in the water column to the gill nets used during 2001. This was done partly because many of the pools which were sampled were too deep, or too rocky, to set the fyke on the bottom. The fyke net was found to be most effective where it could span the width and depth of the river. Generally, fish that were caught in the fyke

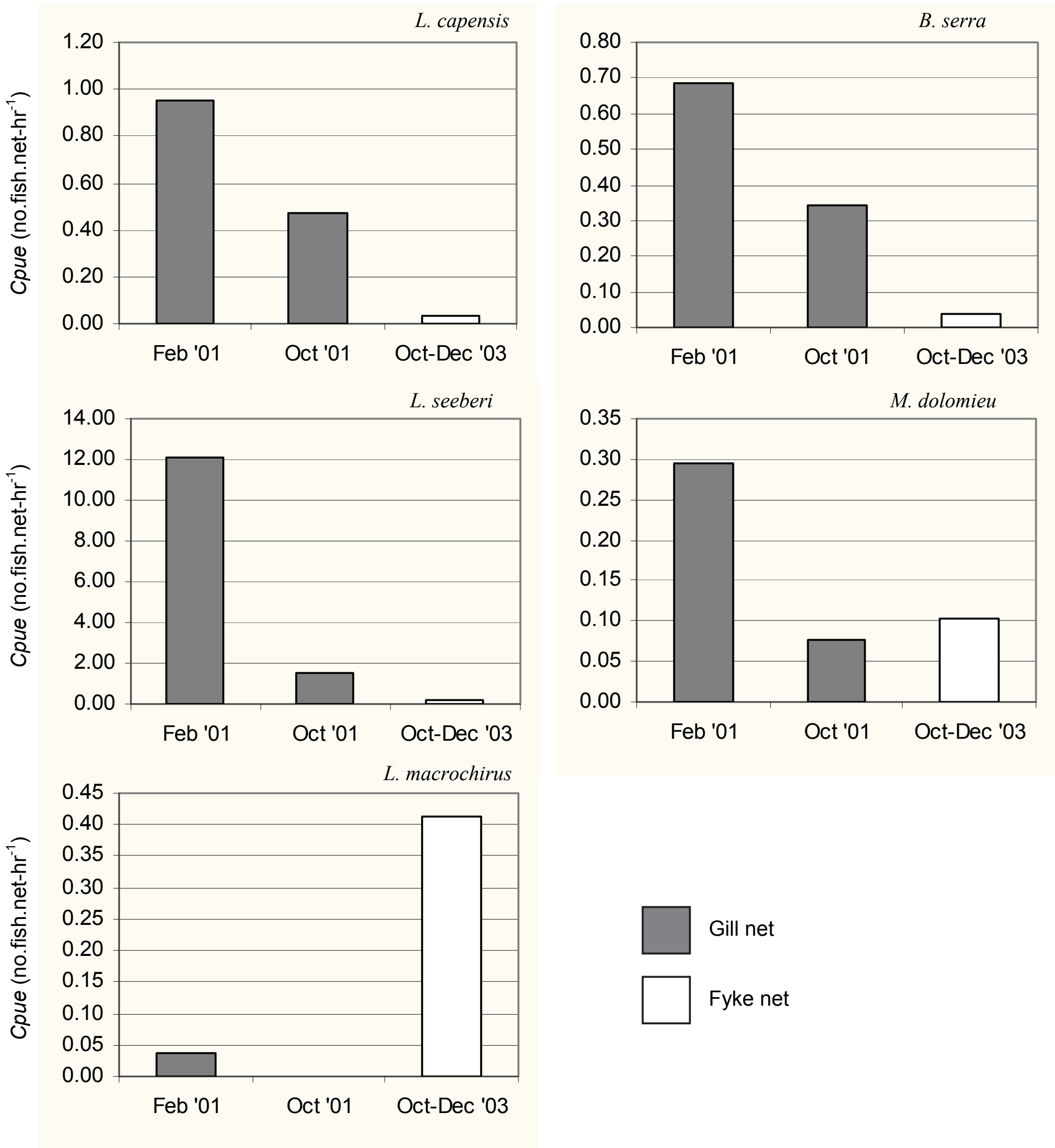

Figure 3.1 Mean catch per unit effort (*cpue*) values for *L. capensis*, (>250 mm TL) *B. serra*, (>250 mm TL) *L. seeberi* (>250 mm TL), *M. dolomieu* (>200 mm TL) and *L. macrochirus* (>150 mm TL) caught in gill nets (February and October 2001) and fyke nets (October-December 2003).

were in far better condition than those caught in the gill nets, although some abrasion of the mucous coating on the surface of the skin and clouding of the eyes was apparent from contact with the sides of the net. Fyke nets are in this respect far superior to gill nets for research on vulnerable fish populations.

An unforeseen disadvantage of using 18 mm mesh in the fyke net was that it gilled the smaller fish. Only a very small mesh size (~3 mm) could eliminate this problem. Increased resistance to strong currents resulting from a smaller mesh size, however, would preclude the use of these nets in reaches with all but the slowest velocities.

In general, the project team felt that fyke nets were far superior to gill nets for working in rivers where endangered species occur, or where the fish need to be returned to the river in good condition. They would therefore be ideal for capturing fish to be used in telemetry studies. However, the low numbers of fish caught with fyke nets precluded their effectiveness where large numbers of fish need to be caught, for example in tagging studies, or where rigorous estimates of abundance are required. Further experimentation in the study rivers would be necessary to determine their sensitivity to detecting changes in abundance. The dimensions of the river channel (width and depth), as well as the behaviour of the fish (which may vary within species between seasons or life stages, or between species) are likely to play a major role in catch variability. These factors would need to be controlled for where more accurate measures of relative abundance are required.

4. CONVENTIONAL TAGGING METHODS

4.1 TAGGING AND MARKING STUDIES IN THE OLIFANTS AND DORING RIVERS

Tagging and marking techniques have been widely used to study fish movements, behaviour, abundance (mark-recapture) and for validation of aging methods (Nielsen *et al.* 1983). Several techniques are available, which vary with respect to their effects on the growth, survival and behaviour of fish, their permanency, the ease with which they can be applied, and the information they convey. The various tagging techniques, together with their advantages and disadvantages have been reviewed by Paxton (2004). The purpose of this section, therefore, is to report on, and evaluate, the suitability of tagging methods that have been used during the course of studies on adult cyprinids in the Olifants and Doring Rivers since 2001. One of the primary aims of the tagging study was to provide empirical evidence for migratory behaviour of endemic fish living in these river systems.

4.1.1. T-bar anchor tags

Individually numbered, medium-sized (~85 mm) Floy® T-bar anchor tags were used to mark the endemic species caught in gill nets during 2001. These tags were approximately 80 mm long (**Plate 4.1**) and were inserted into the musculature of the fish below the dorsal fin by means of a tagging gun. The fish were removed directly from the net before being measured and tagged. There was some concern that the wounds inflicted by the tagging procedure may cause infection, especially in the summer when water quality deteriorates in the standing pools. Another concern was the retention rates of the anchor tags – which may become snagged, or if the wound did not heal properly, would eventually be expelled. During October 2001, when a second survey was undertaken to recapture tagged fish, some indication of the problems associated with using these tags became manifest. Several of the fish captured during these surveys had scars where the tag should have been, suggesting that these had been shed. Two sandfish that had been tagged during February 2001 were recaptured at Rietvlei on the Koebee River and both fish showed evidence of infection where the tag had been inserted. The tag from one of these fish became dislodged while it was being measured. Mortality from infection was also a concern and a more effective means of tagging was therefore sought.

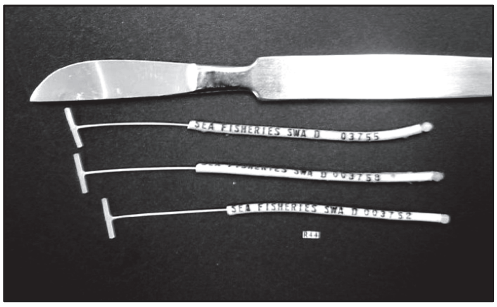

Plate 4.1 T-bar anchor tags used to tag fish during 2001.

4.1.2. Visible Implant Alphanumeric (VI Alpha) tags

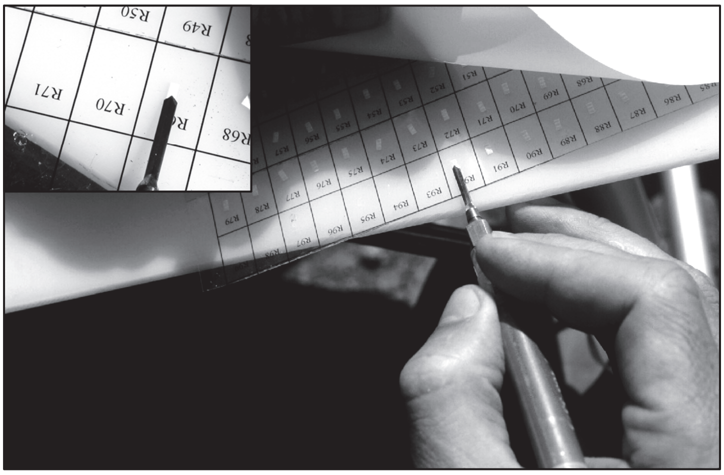

Plate 4.2 Loading the tag into the syringe

Plate 4.3 Inserting the tag between dorsal rays.

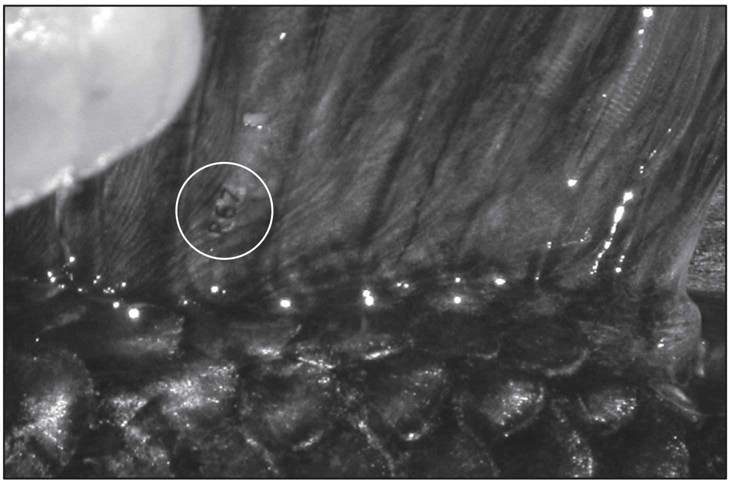

Plate 4.4 Tag inserted between 3rd and 4th dorsal spines.

Soft Visible Implant Alpha numeric (VI Alpha) tags (Northwest Marine Technologies, Inc) were identified as an alternative to the T-bar anchor tag. Soft VI Alpha tags are implanted into transparent adipose tissue (periocular tissue, fin membranes). The tag is implanted by means of a syringe-like injector and fluoresces under a ultra-violet lamp to aid reading. Between September and December 2003, a further ten yellowfish, 16 sawfin and 56 sandfish were marked by means of these tags.

Because of the delicacy of the operation, which involved inserting the tag into the syringe (**Plate 4.2**) and then beneath the tissue of the fish, the net with the fish in was first brought to the shore in the inflatable boat. The fish were anaesthetised individually by immersing them in a bath of 2-phenoxy-ethanol at a concentration of 0.5 ml.l^{-1}. Narcosis was induced after two to five minutes. Once breathing had become irregular and the fish had turned belly-up, it was weighed, measured (TL, FL, SL, girth), tagged and a sample of tissue removed from the inner margin of the pelvic fin for later genetic analyses. All native fish over 300 mm TL were tagged using VI Alpha tags. The tags were inserted beneath the soft adipose tissue at the base of either the 1^{st} and 2^{cnd}, 2^{cnd} and 3^{rd}, or $3^{rd}$and 4^{th} dorsal spines (**Plate 4.3** and **4.4**). Once the fish had recovered in an antiseptic bath (approximately 10 minutes) they were returned to the river.

The VI Alpha tags are less intrusive than the T-bar anchor tags. They left a smaller wound that could be expected to heal more rapidly, and the chances of infection were therefore expected to be considerably less. The VI Alpha tags were, however, more difficult to insert than the T-bar anchor tags, requiring some dexterity in placing the tag at a sufficient depth beneath the surface of the skin to minimise the chances of shedding, but not too deep that the tag number would be occluded by pigmentation in the skin. It was essential therefore that the fish be anaesthetised. An added advantage of anaesthetisation was that it reduced stress by reducing the handling time because the fish was not struggling. This enabled more accurate measurements of length and mass. However, anaesthetisation increases the processing time per fish to between 10 and 15 minutes, thereby reducing the absolute number of fish that could be processed in one day to a maximum of between 20 and 30 large adults. This is in contrast to T-bar anchor tags where up to 80 fish could be processed in little more than an hour if they were removed directly from the net, placed in the inflatable boat and processed on board.

The difficulties of implanting VI Alpha tags, together with the expense and fragility of the applicators, mean that they may not be ideal for extensive application by non-technical staff or recreational anglers. Continued evaluation and experimentation with VI Alpha tags is therefore considered necessary.

4.2 EVALUATION OF THE TAGGING PROGRAMME

The numbers of native fish over 300 mm TL caught and tagged on the Doring River during 2003, together with the site and date on which they were caught, as well as their length specifications and mass are reported in **Table 4.1**. A total of four yellowfish, 12 sawfin and 63 sandfish were tagged. None of these fish have been recaptured in subsequent surveys.

The mark-recapture programme has been ongoing in the catchment since 2001, under the combined support of Department of Water Affairs and Forestry, PGWC and WRC. The total numbers of both native and non-native fish caught between October 2001 and December 2003 in fyke nets, gill nets and seine nets are shown in **Table 4.2**. Native species represented 26 % (1201) of the total number of fish caught (4602). During the course of this programme there have been only three recaptures: one sandfish tagged at Rietvlei on the Koebee River in February 2001 was recaptured at the same site during October 2001; and two sandfish tagged at Aspoort on the Doring River in February 2001 were recaptured at the same site during October 2003.

Table 4.1 Native fish tagged during 2003, together with location, date and tag numbers. Latitude (Lat) and Longitude (Long) are reported in decimal degrees, Total Length (mm TL), Fork Length (mm TL), Standard Length (mm SL), Girth (mm), Mass (g).

SPECIES	SITE		DATE	LAT	LONG	TL	FL	SL	GIRTH	MASS	TAG NO.
Labeobarbus capensis	Melkbosrug	Mr	2003-9-15	-31.859	18.9833	700	627.0	551.0	426.0	4562.0	R58
Labeobarbus capensis	Melkbosrug	Mr	2003-9-17	-31.8590	18.9833	563	499.0	429.0	318.0	1926.0	R48
Labeobarbus capensis	Melkbosrug	Mr	2003-9-17	-31.8590	18.9833	588	524.0	457.0	326.0	2241.0	R47
Labeobarbus capensis	Melkbosrug	Mr	2003-9-17	-31.8590	18.9833	538	473.0	414.0	301.0	1703.0	R46
Labeo seeberi	Uitspanskraal	Uk	2003-9-5	-32.0410	19.4190	525	466.0	410.0	273.0	1349.0	R94
Labeo seeberi	Uitspanskraal	Uk	2003-9-5	-32.0410	19.4190	523	453.0	379.0	281.0	1410.0	R93
Labeo seeberi	Uitspanskraal	Uk	2003-9-5	-32.0410	19.4190	530	469.0	911.0	280.0	1511.0	R95
Labeo seeberi	Uitspanskraal	Uk	2003-9-5	-32.0410	19.4190	455	394.0	350.0	233.0	1082.0	R97
Labeo seeberi	Uitspanskraal	Uk	2003-9-5	-32.0410	19.4190	529	460.0	398.0	289.0	1493.0	R98
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	570	501.0	435.0	319.0	1981.0	R92
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	529	464.0	389.0	296.0	1251.0	R90
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	539	479.0	407.0	292.0	1465.0	R89
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	544	467.0	393.0	270.0	1325.0	R88
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	532	468.0	385.0	282.0	1466.0	R87
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	522	447.0	373.0	276.0	1349.0	R86
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	533	466.0	397.0	298.0	1581.0	R85
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	519	465.0	400.0	276.0	1355.0	R84
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	512	449.0	386.0	279.0	1323.0	R83
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	517	448.0	381.0	270.0	1305.0	R82
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	534	471.0	401.0	299.0	1546.0	R81
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	309	277.0	242.0	157.0	296.0	R80
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	528	453.0	385.0	252.0	1129.0	R79
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	456	394.0	330.0	279.0	1122.0	R78
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	537	458.0	388.0	256.0	1275.0	R77
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	492	433.0	370.0	242.0	1001.0	R76
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	514	450.0	381.0	283.0	1357.0	R75
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	562	498.0	419.0	300.0	1733.0	R74
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	503	444.0	371.0	255.0	1108.0	R73
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	554	498.0	417.0	299.0	1718.0	R72
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	534	477.0	403.0	285.0	1508.0	R71
Labeo seeberi	Uitspanskraal	Uk	2003-9-6	-32.0410	19.4190	504	446.0	377.0	282.0	1297.0	R70
Labeo seeberi	Langkuil	Lk	2003-9-12	-32.0200	19.3079	566	491.0	414.0	259.0	1544.0	R65
Labeo seeberi	Langkuil	Lk	2003-9-12	-32.0200	19.3079	451	395.0	322.0	273.0	1163.0	R64
Labeo seeberi	Langkuil	Lk	2003-9-12	-32.0200	19.3079	530	466.0	393.0	280.0	1426.0	R63
Labeo seeberi	Langkuil	Lk	2003-9-12	-32.0200	19.3079	556	480.0	401.0	298.0	1632.0	R62
Labeo seeberi	Langkuil	Lk	2003-9-13	-32.0200	19.3079	564	484.0	411.0	323.0	1822.0	R61
Labeo seeberi	Langkuil	Lk	2003-9-13	-32.0200	19.3079	537	469.0	397.0	254.0	1257.0	R60
Labeo seeberi	Langkuil	Lk	2003-9-13	-32.0200	19.3079	535	474.0	403.0	225.0	1051.0	R59
Labeo seeberi	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	540	475.0	407.0	259.0	1359.0	R55
Labeo seeberi	Bruinkrans	Bk	2003-10-3	-31.8630	18.8376	576	508.0	470.0	272.0	1620.0	R30
Labeo seeberi	Rietvlei	Rv	2003-11-6	-31.5810	19.0716	445	375.0	349.0	240.0	875.0	R25
Labeo seeberi	Rietvlei	Rv	2003-11-6	-31.5810	19.0716	510	445.0	412.0	254.0	1189.0	R24
Labeo seeberi	Rietvlei	Rv	2003-11-6	-31.5810	19.0716	466	403.0	376.0	230.0	868.0	R23
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	497	433.0	402.0	262.0	1148.0	R21
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	485	438.0	410.0	238.0	1055.0	R20
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	439	379.0	356.0	218.0	730.0	R18
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	478	423.0	392.0	227.0	898.0	R17
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	481	423.0	395.0	232.0	956.0	R16
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	472	410.0	381.0	208.0	780.0	R15
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	473	415.0	389.0	246.0	1040.0	R14
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	483	417.0	387.0	215.0	831.0	R13
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	490	428.0	401.0	225.0	954.0	R12
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	459	390.0	360.0	245.0	985.0	R11
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	521	456.0	427.0	245.0	1146.0	R10
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	468	406.0	375.0	222.0	798.0	R09
Labeo seeberi	Rietvlei	Rv	2003-11-7	-31.5810	19.0716	514	449.0	422.0	251.0	1203.0	R08
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	493	423.0	395.0	232.0	964	R07
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	509	449.0	415.0	228.0	1040	R06
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	518	447.0	415.0	230.0	1015	R05
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	485	425.0	396.0	236.0	978	R04
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	466	405.0	376.0	222.0	824	R03
Labeo seeberi	Rietvlei	Rv	2003-11-8	-31.5810	19.0716	467	410.0	384.0	220.0	845	R02
Labeo seeberi	Ondertuin	Ot	2003-11-13	-31.5810	19.0716	534	459.0	425.0	265.0	1338	R01
Labeo seeberi	Oudrif	Od	2003-12-5	-31.8570	18.9135	571	497.0	459.0	276.0	1654.0	W08
Labeo seeberi	Oudrif	Od	2003-12-5	-31.8570	18.9135	534	472.0	435.0	271.0	1484.0	W09
Labeo seeberi	Langkuil	Lk	2003-12-10	-32.0190	19.3077	545	492.0	457.0	270.0	1575.0	W12
Labeo seeberi	Langkuil	Lk	2003-12-10	-32.0190	19.3077	541	468.0	434.0	285.0	1326.0	W13
Barbus serra	Oudrif	Od	2003-9-9	-31.8570	18.9135	450	386.0	337.0	238.0	855.0	R67
Barbus serra	Oudrif	Od	2003-9-9	-31.8570	18.9135	462	402.0	349.0	245.0	965.0	R66
Barbus serra	Melkbosrug	Mr	2003-9-15	-31.8590	18.9833	484	426.0	374.0	306.0	1426.0	R56
Barbus serra	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	446	387.0	331.0	241.0	953.0	R54
Barbus serra	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	439	377.0	322.0	262.0	975.0	R53
Barbus serra	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	443	390.0	331.0	234.0	830.0	R52
Barbus serra	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	450	382.0	334.0	260.0	984.0	R51
Barbus serra	Melkbosrug	Mr	2003-9-16	-31.8590	18.9833	446	388.0	333.0	256.0	1055.0	R49
Barbus serra	Bruinkrans	Bk	2003-10-3	-31.8630	18.8376	473	429.0	397.0	264.0	1165.0	R29
Barbus serra	Bruinkrans	Bk	2003-10-3	-31.8630	18.8376	511	456.0	425.0	270.0	1283.0	R26
Barbus serra	Bruinkrans	Bk	2003-10-3	-31.8630	18.8376	440	391.0	365.0	232.0	1852.0	R27
Barbus serra	Rietvlei	Rv	2003-11-6	-31.5810	19.0716	264	229.0	215.0	108.0	181.0	R22

Table 4.2 Summary of the total numbers of native and non-native fish caught, tagged and recaptured (recap = recaptured) throughout the Olifants and Doring Rivers during 2001, 2002, 2003, as well as the numbers of non-native species recorded over the same period. The table combines adult and juvenile fish of all species. Those fish which were not tagged were either too small, or were kept for biological examination.

	2001		2002		2003		Total		Recap
Species	**Caught**	**Tagged**	**Caught**	**Tagged**	**Caught**	**Tagged**	**Caught**	**Tagged**	
L. capensis	54	45	4	0	16	4	74	61	0
B. serra	282	31	7	0	204	12	493	44	0
L. seeberi	453	304	75	0	83	63	611	371	3
B. anoplus	23	0	0	0	0	0	23	0	0
Total	**812**	**380**	**86**	**0**	**303**	**79**	**1201**	**465**	**3**
M. dolomieu	131	0	4	0	257	0	392	0	0
M. punctulatus?	18	0	0	0	0	0	18	0	0
M. salmoides	3	0	0	0	0	0	3	0	0
L. macrochirus	1639	0	1	0	1278	0	2918	0	0
T. sparrmanii	41	0	0	0	21	0	62	0	0
O. mossambicus	8	0	0	0	0	0	8	0	0
Total	**1840**	**0**	**5**	**0**	**1556**	**0**	**3401**	**0-**	**0**

The reason for the paucity of data from this programme can be ascribed to the following:

- *sampling was not continued over a sufficiently long period of time* – mark-recapture programmes need to be continued over several years;
- *low capture and tagging rates* – the absolute abundance of fish in the mainstem of the rivers is low;
- *sampling was not intensive enough* – increased effort per segment and site, as well as increased manpower, would be necessary for the tagging programme to yield appreciable amounts of data.

Because of the low recapture rates, tagging programmes are not considered practical for studying the seasonal movements of riverine species in large river systems with limited manpower. This is particularly true where population numbers are low and where there are few commercial or recreational fishers who could contribute to the tagging programme. For mark-recapture studies to be sustainable in the Olifants and Doring Rivers, it is here recommended that they be continued over a longer period (>5 years), that the studies be more contained, (i.e. a shorter river segments), and that they be administered by conservation bodies, preferably in collaboration with recreational fishers.

5. TELEMETRY PILOT STUDY

5.1. INTRODUCTION

Given the difficulties and limitations of tracking fish movement in the study rivers using conventional tagging methods, it has become clear that telemetry is the most viable alternative. One of the primary objectives of the current study was therefore to assess the feasibility of using telemetric techniques to study the spatial behaviour of cyprinids in the Doring River.

During 2002 a research programme was planned with telemetry specialists from the Norwegian Institute of Nature Research (NINA). This programme, involving collaboration between UCT, the South African Institute for Aquatic Biodiversity (SAIAB) and NINA, would require that radio telemetry transmitters with a minimum of 12 or 24 month lifespan be implanted in 30 Clanwilliam yellowfish, 30 Clanwilliam sandfish, and 30 sawfin, all taken from the Doring River. The position of each fish would be ascertained every two weeks by tracking them by aircraft between May and December – these being the months when the fish are likely to be most active. Aircraft tracking would be supplemented by more intensive ground-based tracking during critical periods, i.e. after the first rains of the high flow season (May – July) and over the spawning season (October – December). Before this study could proceed, it was agreed that pilot studies needed to be conducted to ascertain the sensitivity of the study species to the implantation of transmitters. The mortality rate and healing process in Clanwilliam yellowfish, sawfin and sandfish fitted with dummy radio transmitters, was therefore monitored in captive wild fish during 2003.

5.2. METHODS

During September 2003, eight adult yellowfish, 11 sandfish and one sawfin were collected from Aspoort and De Mond on the Doring River and transported back to Cape Town with the assistance of staff and vehicles from the Two Oceans Aquarium. The fish were transported in a 1000 *l* trailer designed specifically for the purpose. Water changes were conducted daily while the fish were in the tank. The water was aerated continuously with pure oxygen which has a sedative effect on the fish. Low dosages of anaesthetic (2-phenoxy-ethanol) were added to the water to calm the fish after it was discovered that sandfish were jumping against the side of the tank. Despite these precautions eight sandfish were lost during transportation to Cape Town. The remaining 12 fish were held at the Two Oceans Aquarium (six yellowfish, two sandfish) and UCT (two yellowfish, one sandfish, one sawfin).

Dummy transmitters were inserted into the captive fish. The dummy transmitters were fitted with one of two types of antennae: (1) coiled antennae, where both the transmitter and antenna are completely encapsulated in resin and held within the body of the fish and, (2) whip antennae, which protrude from

Plate 5.1 A 20-30 mm incision on the ventral surface is opened into the body cavity of the fish.

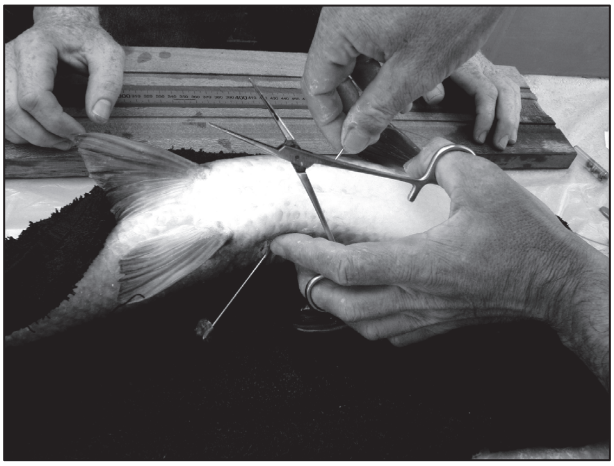

Plate 5.2 A separate opening is made for the whip antenna.

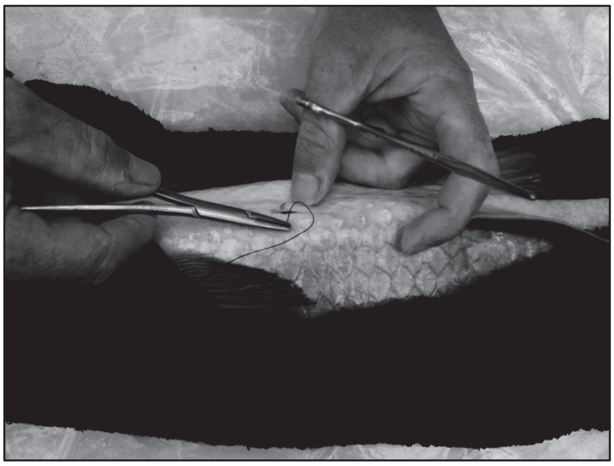

Plate 5.3 The dummy transmitter is inserted and the wound is sutured closed.

the resin and exit the body of the fish. Whip antennae have a higher field reception range than coiled antennae, but there is the danger that they may snag, or irritate the exit wound.

The fish were first anaesthetised in the manner described in the previous section. Before surgery, each fish was weighed and measured. A 20 - 30 mm incision was made parallel to the midline of the ventral surface halfway between the pectoral and pelvic fins (**Plate 5.1**). The dummy transmitter was then inserted into the body cavity. In the case of fish fitted with transmitters which had whip antennae, a separate opening was made with a hypodermic needle through which the antenna was then extended (**Plate 5.2**). The wound was closed with three interrupted sutures (**Plate 5.3** and **5.4**). Transmitters were implanted in ten fish: whip antennae in four yellowfish, one sawfin and two sandfish and coiled antennae in the remaining two yellowfish and sandfish. The fish ranged in size from 450 – 630 mm TL.

Aquarium facilities at the university of Cape Town consisted of a 2000 *l* tank with a discontinuous filtration system comprising a 1000 *l* holding tank and a 1000 *l* gravel-bed filter. Stocking densities in this facility were approximately 300 *l*/fish. The aquarium facilities at the Two Oceans Aquarium consisted of a 2000 *l* tank with a continuous filtration system. Specimens were retained for an acclimation period of one week prior to surgery.

Plate 5.4 Fin Okland (NINA) sutures closed a Clanwilliam yellowfish after inserting a whip antenna. The antenna can be seen protruding from the behind the anal fin.

5.3. RESULTS

The details of the study animals implanted with dummy transmitters are reported in **Table 5.1**. All but two of the study animals died between October and December 2003. The loss of these animals, however, was attributed to infections and diseases resulting from their confinement rather than the surgery or transmitter. The loss of the majority of sandfish during their transportation suggested that these fish are particularly sensitive to stress, possibly induced by water-quality deterioration and/or confinement. It was suggested that the 1000 *l* transport trailer had been stocked with too many fish resulting in a build up of urea, despite the regular water changes. Poor water-quality at the UCT aquaria as a consequence of an inadequate filtration system resulted in the fish being kept here eventually succumbing to fungal and anchor worm (*Lernea*) infections. All the fish kept at UCT died in early November after attempts to treat both the fungal and anchor worm infections failed. Until December, the fish kept at the Two Oceans Aquarium recovered well from the surgery and the wounds were healing well. Three months after the surgery, however, in December 2003, a protozoan infection (*Ichthyophthirius*) killed all but two of the yellowfish. These remaining yellowfish are currently (2004) being held at the Jonkershoek fish hatchery in Stellenbosch, Western Cape. A post mortem examination of the wounds in the dead fish from the Two Oceans Aquarium suggested that before their deaths, the surgery wounds had closed.

Table 5.1. List of fish implanted with dummy radio transmitters during October 2003. The first tag number reports the number of the VI Alpha tag, the second tag number reports the number of the dummy transmitter.

Species	TL (mm)	FL (mm)	SL (mm)	Mass (g)	Tag no.	Tag no.	Type	Location	Cause of death
Barbus serra	530.00	520.0	468.0	2000.0	R32	H-37	whip	UCT	Infection
Labeo seeberi	522.00	458.0		1200.0		L-6	whip	Two Oceans	White spot
Labeo seeberi	534.00	492.0		1200.0	R43	L-12	coil	Two Oceans	White spot
Labeo seeberi	510.00	450.0	410.0	1400.0	R33	H-45	whip	UCT	Infection
Labeobarbus capensis	652.00	602.0	566.0	2850.0	R42	T	whip	Two Oceans	White spot
Labeobarbus capensis	708.00	630.0	578.0	4000.0	R39	L1	whip	Two Oceans	White spot
Labeobarbus capensis	640.00	586.0	558.0	2650.0	R38	L10	whip	Two Oceans	(alive 2004)
Labeobarbus capensis	524.00	460.0	432.0	1400.0	R36	11	coil	Two Oceans	(alive 2004)
Labeobarbus capensis	522.00	462.0	446.0	1400.0	R37	L3	coil	Two Oceans	White spot
Labeobarbus capensis	570.00	520.0	490.0	1900.0	R35	H	whip	Two Oceans	White spot

5.4. CONCLUSION

Despite the deaths of all but two fish, the team felt that the experiment had been worthwhile. Should funding become available for the telemetry study to proceed, wild fish would be captured from the Doring River, the transmitters inserted, and the fish returned immediately to the river instead of being transported to Cape Town – the worst possible scenario.

6. FISH DISTRIBUTION IN THE LOWER OLIFANTS AND DORING RIVERS

6.1. INTRODUCTION

The results of the 2003 surveys are reported in this chapter and discussed in the light of previous surveys. Due to the low numbers of native fish caught, the comparatively few replicates and therefore high variability of the dataset, only a qualitative interpretation of the abundance and distribution of species has been possible. No attempt has been made to analyse or represent the data statistically.

The focus was on the lower Olifants and Doring Rivers where the proposed water-resource developments would take place. Additional funds from the WODRIS study (PGWC 2004) enabled the scope of the 2003 WRC project to be extended, and information acquired during the course of these surveys provided the basis for low-confidence predictions on the impacts of a dam on fish populations in the lower Doring River.

6.2. FISH ABUNDANCES AND DISTRIBUTION

Tables 6.1 and **6.2** report fyke net catches during May, and September to December 2003, respectively. The fish caught during 2003 (**Table 6.2)** have been separated into juveniles and adults. Size at maturity (L_{50}) was estimated on the basis of best available knowledge since, even where reliable estimates are available for the bass and bluegill, life history parameters in local systems are likely to differ from those reported for their country of origin. Size at maturity of smallmouth bass at a maximum size (L_{inf}) of 520 mm was reported by Fishbase (Froese and Pauly 2004) to be 224 mm TL. Smallmouth bass as small as 200 mm TL were ripe and running in the Doring River, however, and this size was therefore set as the length of mature fish. Size at maturity of bluegill sunfish at a maximum size (L_{inf}) of 250 mm TL was reported by Fishbase to be 150 mm which corresponded to lengths of ripe and running fish in the Doring River. Jubb estimated L_{50} of Clanwilliam yellowfish at 250 mm TL and this size was taken for sawfin as well. Moggel *Labeo umbratus* and Orange River Labeo *Labeo capensis* have been reported to mature at 330-400 mm TL (Allanson and Jackson 1983) and so 350 mm TL was take as L_{50} for Clanwilliam sandfish.

During May 2001, bluegill sunfish comprised an overwhelmingly high proportion (82 %) of the catch in the lower Olifants and Doring Rivers (**Table 6.1**). These were caught in the small fyke nets at all the sites between the Bulshoek Barrage and the estuary. The highest proportion of the catch came from Draairivier (*Dr*) on the Olifants and Oudrif (*Od*) (**Figure 2.1**) on the Doring River. The fyke nets were set in flowing water at Lutzville (*Lv*), which may account for their absence in catches from here.

Table 6.1. Species and number of fish caught during the course of surveys conducted in the Olifants and Doring Rivers during May 2003 using the small fyke nets. OR = Olifants River, DR = Doring River

River	Sites	Species				
		L. macrochirus	*Micropterus spp.*	*O. mossambicus*	*T. sparrmanii*	*M. cephalus*
Olifants	Kleinrietvlei *Kr*	54	0	0	0	0
Olifants	Sandkamp *Sk*	14	0	0	0	0
Olifants	Kransgat *Kg*	11	19	3	14	
Olifants	Gideonsoord *Go*	2	1	0	9	1
Olifants	Draairivier *Dr*	348	3	35	16	23
Olifants	Lutzville *Lv*	0	0	0	2	1
Doring	Bruinkrans *Bk*	16	3	0	0	0
Doring	Oudrif *Od*	184	4	0	0	0
	Total	***629***	***20***	***38***	***41***	***25***

The co-occurrence of both smallmouth and spotted bass *Micropterus punctulatus* in the reaches between Bulshoek and the Olifants River estuary has complicated the identification of juvenile and young adult bass in the system. In addition, hybridisation between these two species is known to occur (Koppelman 1994) and since some fish appeared to have both spotted and smallmouth characteristics, all bass have been designated *Micropterus* spp. Apart from bluegill sunfish, banded tilapia *Tilapia sparmanii* and Mozambique tilapia *Oreochromis mossambicus* were most commonly caught in these reaches. Flathead mullet *Mugil cephalus* were caught as far upstream as Draairivier, approximately 46 river-km from the Olifants River mouth.

The complete absence of the native cyprinids in the lower Olifants River catches could be attributable to gear selectivity. The study team feels, however, that populations of native species here are extremely small and localised. Gill nets have proved effective for catching the cyprinids (Paxton *et al*. 2002), but despite their application during the 2001 surveys, only two yellowfish have been caught in 8.5 net-hrs (Paxton *et al*. 2002). In addition, dive surveys downstream of Bulshoek confirmed the presence of only bass and bluegill sunfish. Interviews with farmers and the members of the Lutzville angling club confirmed that yellowfish are extremely rare and confined to the Cascade Pools region. Most of those interviewed had never heard of a sawfin or sandfish, but a few remembered having seen sandfish in the Olifants River prior to the 1970s. It is likely therefore that Clanwilliam yellowfish have been reduced to small and isolated populations, and that sawfin and sandfish have become locally extinct in the Olifants River in the last few decades. It is suggested that a systematic questionnaire survey may yield more information regarding the past and present status of native fish species in this region than ecological surveys.

On the Doring River at Oudrif (*Od*) and Bruinkrans (*Bk*) during 2003, schools of between 20 and 30 adult fish belonging to one of the native cyprinid species (Clanwilliam yellowfish or sawfin) were observed from boats and from the banks. Despite extensive trials with the small fykes at both of these sites, however, the smaller nets failed to capture the indigenous species (the reasons for this are addressed in Chapter 3). **Table 6.2** reports the numbers of fish caught in the Doring River later during 2003 (September – December) using the larger custom built fyke net. The sites where each of the species were caught together with an indication of the relative proportions at each site, are shown in **Figures 6.1 – 6.5**.

Adult native cyprinids comprised roughly 40 % (83) of the total catch during the course of the 2003 surveys, and these were primarily sandfish caught at Uitspanskraal (Uk) (27 adults) near the Biedouw River confluence on the Doring River and at Rietvlei (Rv) on the Koebee River (22 adults) located in the northernmost reaches of the Doring River catchment (**Figure 6.5**). Both these sites have yielded large numbers of sandfish in past surveys (Paxton *et al*. 2002). Significantly, these sites are located in reaches where the local topography is dominated by Bokkeveld shales. For approximately 60 river km, between the confluence of the Bos River and Doringbos, the Doring River flows through shales and mudstones of the Karoo Series. The river has eroded laterally here, in contrast to the vertical erosion where it flows through the more resistant quartzitic sandstones of the Table Mountain Series (TMS). Meandering sandbed pools characterise the shale zones, whereas bedrock rapids and runs are more common in TMS zones. Sandfish therefore appear to favour the sandbed pools of the middle reaches of the Doring River from the confluence of the Bos River to Doringbos and are found in large numbers in similar pools in the Koebee River. Both yellowfish and sawfin are caught less frequently in these regions.

Table 6.2. Species and numbers of juvenile and adult native and non-native fish caught in the Doring River system during the course of surveys conducted in September (09), October, (11) and December (12) 2003 using the large fyke net. DR = Doring River, OK = Oorlogskloof River, KB = Koebee River.

				NATIVE									**NON-NATIVE**					
	SITE			*L. capensis*			*L. seeberi*			*B. serra*			*Micropterus* spp.			*L. macrochirus*		
			Mon	Juv	Adult	**Tot**	Juv	Adult	**Tot**	Juv	Adult	**Tot**	Juv	Adult	**Tot**	Juv	Adult	**Tot**
DR	Bruinkrans	**Bk**	10	0	0	**0**	0	1	**1**	0	3	**3**	35	0	**35**	84	21	**105**
			12	0	0	**0**	0	0	**0**	0	2	**2**	0	2	**2**	52	26	**78**
DR	Langkuil	**Lk**	09	0	0	**0**	0	7	**7**	0	0	**0**	0	1	**1**	0	1	**1**
			12	0	0	**0**	0	2	**2**	0	0	**0**	3	1	**4**	52	3	**55**
DR	Melkboom	**Mb**	09	0	0	**0**	0	0	**0**	0	0	**0**	0	1	**1**	2	1	**3**
			11	0	0	**0**	1	0	**0**	0	0	**0**	1	1	**2**	22	10	**32**
DR	Melkbosrug	**Mr**	09	0	4	**4**	0	1	**1**	0	6	**6**	0	6	**6**	2	3	**5**
			12	0	0	**0**	0	0	**0**	0	2	**2**	1	14	**15**	55	46	**101**
DR	Oudrif	**Od**	09	0	0	**0**	0	0	**0**	0	2	**2**	0	0	**0**	9	10	**19**
			12	0	0	**0**	1	2	**3**	0	0	**0**	3	2	**5**	162	20	**182**
DR	Uitspanskraal	**Uk**	09	0	0	**0**	0	27	**27**	0	0	**0**	0	0	**0**	0	0	**0**
			12	0	0	**0**	0	0	**0**	0	0	**0**	16	0	**16**	45	4	**49**
OK	Brakwater	**Bw**	11	0	0	**0**	3	0	**3**	187	0	**187**	0	0	**0**	0	0	**0**
KB	Rietvlei	**Rv**	11	0	0	**0**	0	22	**22**	0	1	**1**	3	4	**7**	247	7	**254**
KB	Ondertuin	**Ot**	11	0	0	**0**	0	1	**1**	0	0	**0**	0	3	**3**	135	6	**141**
	Total			***0***	***4***	***4***	***4***	***63***	***68***	***187***	***16***	***203***	***62***	***35***	***97***	***867***	***158***	***1025***

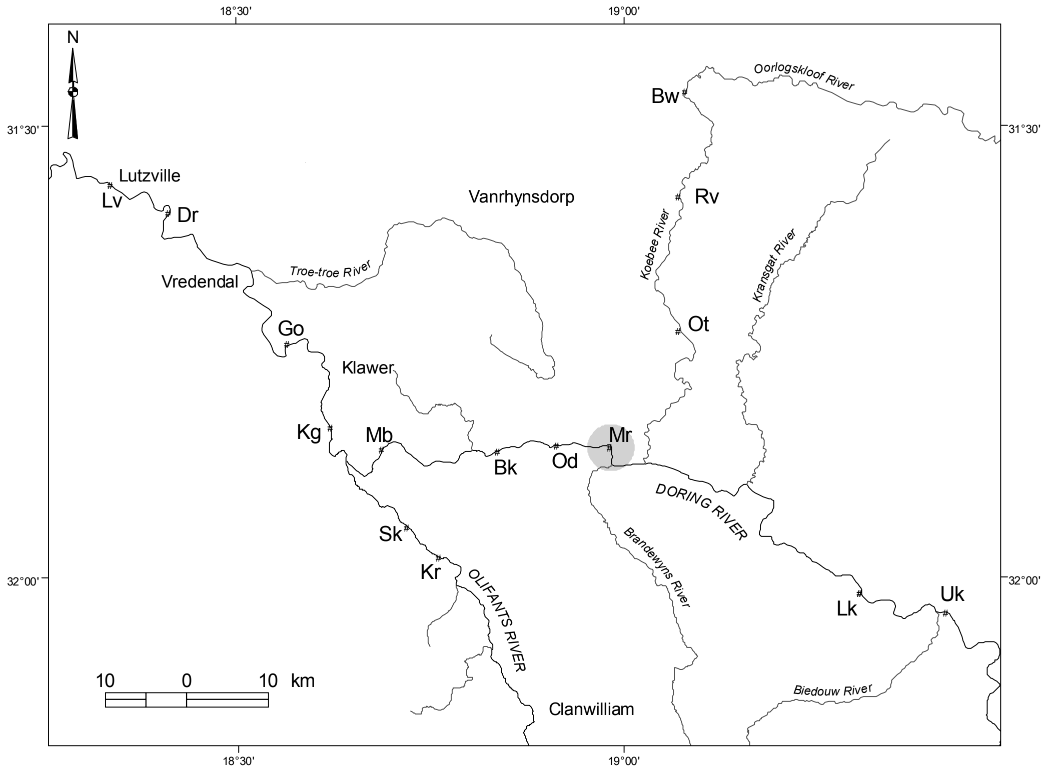

Figure 6.1 Map of the 2003 study sites (black dots) showing the occurrence of Clanwilliam yellowfish caught at each site (shaded circles). The size of each shaded circle is proportional to the number of fish caught at the site.

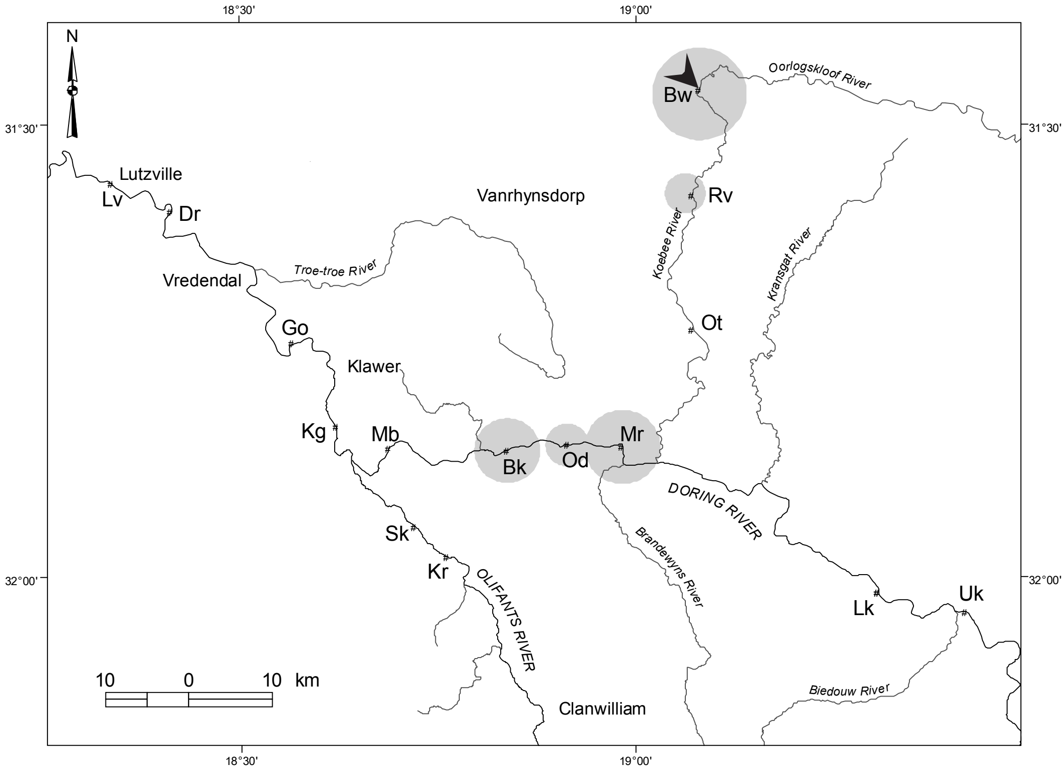

Figure 6.2 Map of the 2003 study sites (black dots) showing the occurrence of sawfin (shaded circles). The size of each shaded circle is proportional to the number of fish caught at the site.
➤ indicates sites where juveniles were caught.

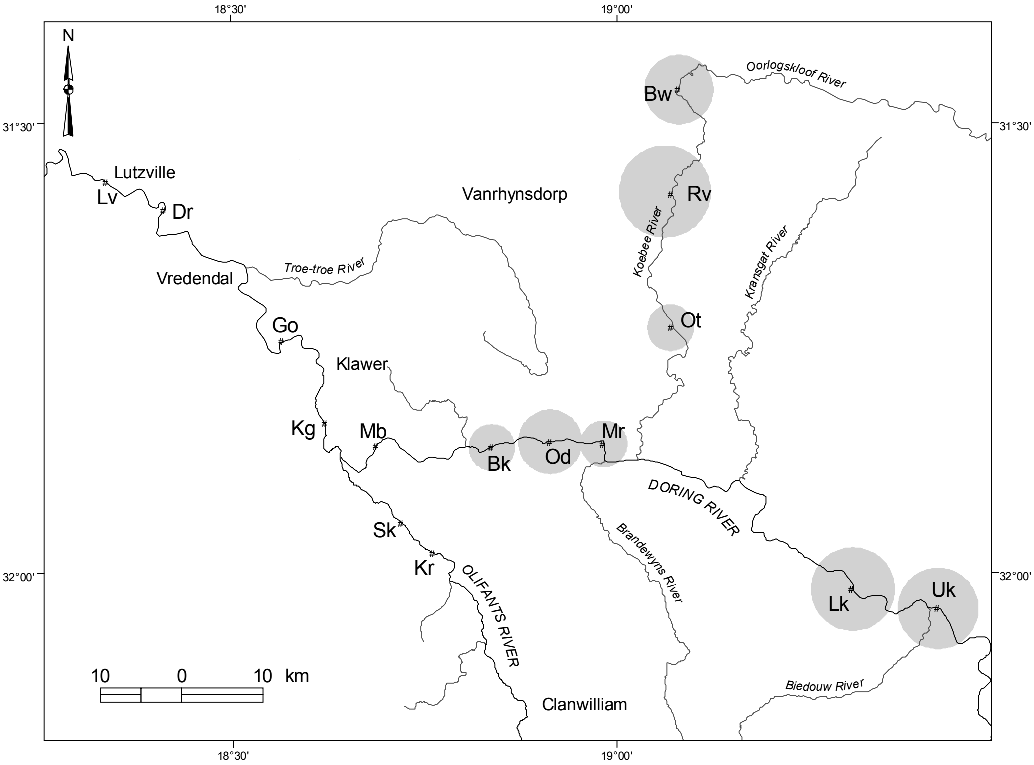

Figure 6.3 Map of the 2003 study sites (black dots) showing the occurrence of Clanwilliam sandfish (shaded circles). The size of each circle is proportional to the number of fish caught at the site.

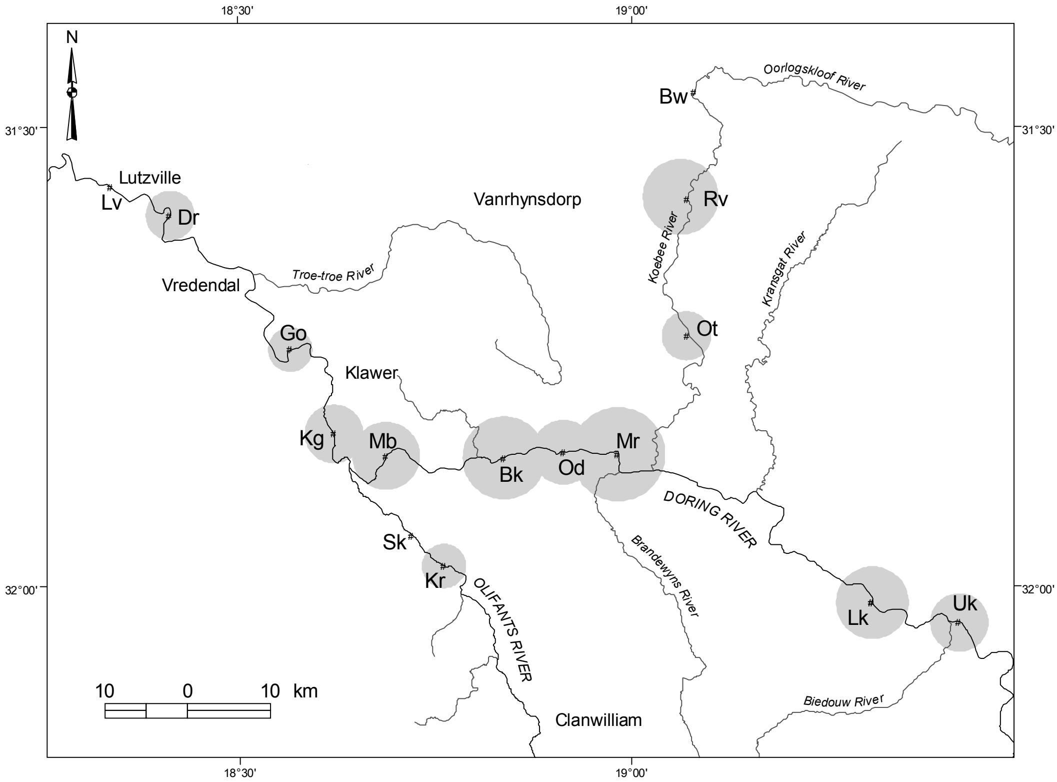

Figure 6.4 Map of the 2003 study sites (solid black circles) showing the occurrence of bass (shaded circles). The size of each circle is proportional to the number of fish caught at the site.

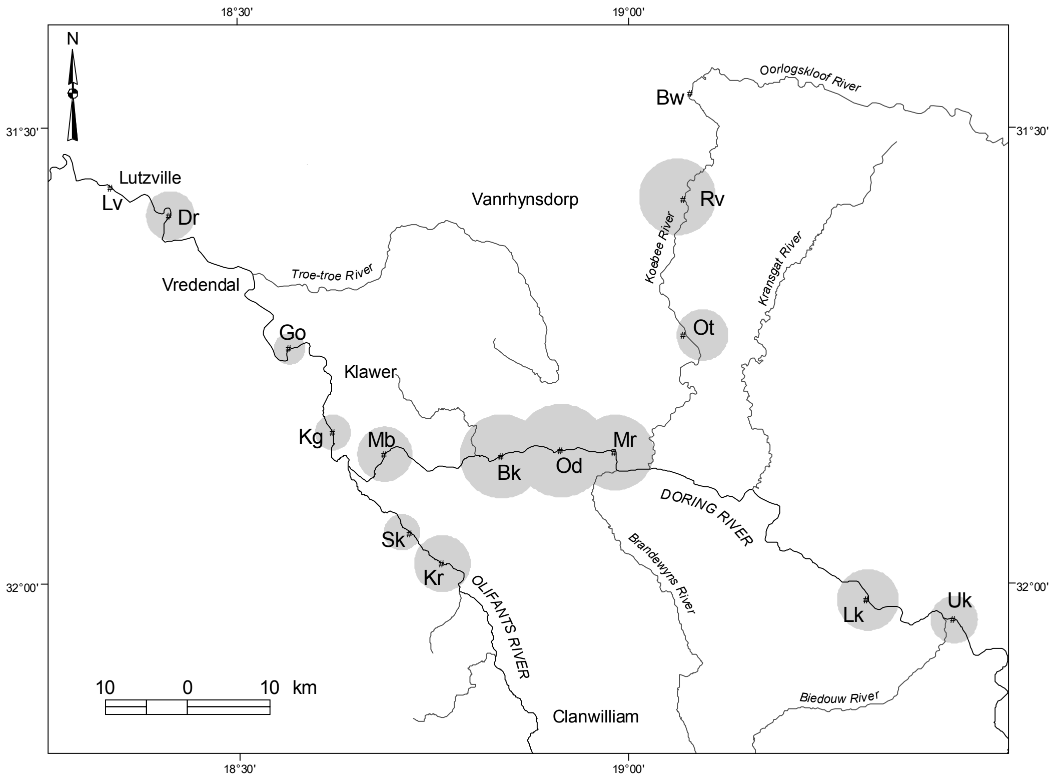

Figure 6.5 Map of the 2003 study sites (black dots) showing the occurrence of bluegill sunfish (shaded circles). The size of each circle is proportional to the number of fish caught at the site.

Yellowfish and sawfin appear to favour river segments which are bedrock-controlled resistant standstone reaches. Catches of adults of both these species, although low (four and 16 respectively), were made at Bruinkrans *Bk*, Melkbosrug *Mr* and Oudrif *Od*.

In total, sawfin were the most abundant, with adult native fish being caught between September and December 2003. This has confirmed the results of previous surveys, which suggest that sawfin occur in the lower Doring River downstream of the Kransgat River in greater numbers than elsewhere in the catchment (Paxton *et al*. 2002 and Paxton 2004) (see **Figure 6.2**). They appear to favour the deep runs and bedrock controlled rapids in the lower Doring River, which flows through resistant sandstone in this region.

The exception to the occurrence of sawfin in the lower Doring River is the high numbers of juveniles at Brakwater *Bw* on the Oorlogskloof River. The river here is no more than 5 m wide, but has been found to support large numbers of juvenile sawfin. Similar nursery refuges free of invasion by bass and bluegill sunfish can be found throughout the catchment in tributaries of both the Olifants and Doring Rivers. The almost complete absence of juvenile fish in any mainstem samples since the sampling began in 2001 suggests that the invasive species are causing catastrophic recruitment failures outside the tributaries.

NOTE: the high numbers of juvenile sawfin caught at Brakwater (Bw) were an unexpected consequence of using 18 mm mesh on the fyke net – this diameter mesh gilled the smaller fish. It is not recommended, therefore, that mesh sizes larger than 3 mm be used for constructing fyke nets for fishing areas where juveniles of a threatened fish species are expected.

The one unusual exception to the absence of juvenile fish in the mainstem reaches was the capture of a single juvenile sandfish (47 mm TL) at Melkboom *Mb* in November 2003. Since the surveys began in 2001, no indigenous fish had been caught at Melkboom, the most downstream site on the Doring River, either by means of gill-nets, fyke-nets or seine nets. The single juvenile sandfish, caught in December 2003, was the first and only indigenous fish recorded in the Doring mainstem through this whole sequence of surveys. Presumably it had originated from a spawning event further upstream. Intensive seining of the same pool did not yield more individuals.

Bass and bluegill sunfish (**Figures 6.4** and **6.5** respectively) were ubiquitous throughout the study area, reflected by their occurrence and high abundance in all catches apart from the Brakwater (BW).

6.3. DISCUSSION

6.3.1. Fish movement between the Olifants and Doring Rivers

Populations of Clanwilliam yellowfish, sawfin and sandfish are known to have occurred in the lower Olifants River below the Bulshoek Barrage in large numbers prior to the 1950s (Harrison 1976). Evidence gathered since 2001, however, shows that the number of indigenous fish in these reaches is now extremely low. On five fish surveys over the past three years, only two yellowfish have been caught in gill-nets in the lower Olifants River downstream of the Bulshoek Barrage and, despite good underwater visibility, none have been observed during dive surveys.

In addition, most farmers, local fishermen and other members of the community who live along the banks of the Olifants River between Klein Reitvlei (*Kr*) and Lutzville (*Lv*), and who were interviewed during the course of the surveys, say they have rarely seen or caught yellowfish, and cannot recall having seen sawfin or sandfish.

Those who do recall having seen yellowfish say that their numbers have dropped substantially since the 1980s. During the earlier half of the twentieth century, large numbers of the endemic species were harvested from pools near the gauging weir at Melkboom (*Mk*), loaded onto wagons, and sold at local markets in Klawer. Farmers who grew up alongside the river, however, last remember seeing large populations of sandfish and yellowfish between 30 – 40 years ago. By contrast, bass, bluegill sunfish and mullet are now well known and caught on a regular basis.

The combination of sample data and anecdotal information suggests that endemic fish that remain in the lower Olifants River are most likely a remnant of a much larger population that may have moved between the Olifants and Doring Rivers. Habitat degradation in the lower Olifants River, such as encroachment of riparian vegetation and reduced connectivity between pools, however, may have considerably reduced the availability of, and access to, spawning sites, and thus limited recruitment to these populations in recent years. The large instream barriers (Clanwilliam Dam and Bulshoek Barrage) may have also prevented recolonisation of these reaches through downstream transport of larvae, or active movement of juveniles and adults from the upstream reaches. Some recruitment may occur from the Doring River, particularly during high flows, but current population sizes suggests that this is minimal. This contention is supported by other information collected during site visits and aerial surveys of the lower Olifants River, such as:

- there are few large pools, which are the preferred habitat of the cyprinids;
- the pools that are there are separated by long, shallow stretches where the river braids through mid-channel sandbanks and riparian vegetation, which would limit fish movement to the high-flow season, and for a brief period thereafter when the pools were sufficiently connected to allow movement;
- the opportunity for migration is diminished still further by flow regulation which reduces the depths in critical habitat such as riffles, rapids and causeways;
- no endemic fish were captured at the most downstream Doring River site sampled, *viz.* Melkboom *Mk* situated upstream of the Melkboom gauging weir, 7 km upstream of its confluence with the Olifants River.

Thus, the conclusion of this study is that the existence of a synchronised seasonal migration by large numbers of endemic fish between the Olifants and Doring Rivers, which may contribute to fish production in both rivers, is highly unlikely.

It is plausible, however, that the individuals in the Olifants River originate from infrequent displacement of individuals from the Doring River during peak flows which, when they occur, may be followed by compensatory upstream movements back upstream into the Olifants as well as into the Doring to spawning grounds in spring (e.g. Lucas and Batley 1996). There are still substantial populations of endemic cyprinids in the mainstem of the Doring River upstream of the Melkboom site (*Mk*). Fish surveys conducted here since 2001 have confirmed that significant populations of yellowfish, sandfish and unusually large numbers of adult sawfin (which are rare in the remainder of the catchment) persist in these reaches. While bass and bluegill predation limit the recruitment success of fish in these reaches, these populations may still occasionally contribute to fish production here.

6.3.2. Proposed dams as barriers to fish movement

The conclusions presented above lead to the following assessments of the likely barrier impacts associated with the water-resource developments that were considered in the WODRIS study.

Dam at Melkboom:	Unlikely to represent a major barrier to fish movement at the current levels of fish in the lower Olifants River.
Dam at Melkbosrug:	Significant populations of yellowfish, sandfish and unusually large numbers of adult sawfin persist in the middle and lower Doring River. A dam located here would represent a barrier to fish movement.
Abstraction weir:	As for a dam at Melkbosrug.

6.4. CONCLUSION

A fishway at Bulshoek Barrage, together with the implementation of the ecological Reserve for the lower Olifants River *may* assist in reinstating some of the links between the lower Olifants River and the Doring River, and increasing the populations of indigenous fish in the lower Olifants River, thereby increasing carrying capacity of the system, and enhancing the chances of reproductive success of *L. capensis*, in particular. However, such a rehabilitation programme would need to be accompanied by frequent large floods and a reduction of vegetation encroachment in order to clear riffle habitats and reinstate connectivity between the pools.

Finally, given the extreme sensitivity of the tributaries, it is suggested that no further water-resource developments take place in any tributaries that are identified as high production units for native species, and that these rather be rehabilitated and designated aquatic protected areas. A more rigorous assessment of key tributaries will need to involve all stakeholders, but a preliminary assessment may include: the Koebee/Oorlogskloof system (sandfish), Biedouw River (sandfish), Matjies/Driehoeks system (yellowfish and sawfin), Rondegat River (yellowfish), Boskloof and Ratels Rivers (yellowfish) as well as the reaches and tributaries of the Olifants River upstream of the farm Keerom (yellowfish and sawfin) as protected areas.

7. CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

The surveys conducted between 2001 and 2003 were driven largely by immediate management concerns. Detecting change and predicting the outcomes of anthropogenic disturbance, however, is difficult where baseline data are limited. There have been limited studies into the ecological requirements of native fish of this system (e.g. Gore *et al*. 1991; Cambray *et al*. 1997; King *et al*. 1998), and no studies on their interactions with introduced invasive species. Confidence in the management recommendations that have been made in this report and in the WODRIS study (PGWC 2004) on the basis of 'best available knowledge' is therefore extremely low. While the current surveys have given some indication of the distribution and size-structures of fish in the catchment, they have provided few data upon which to develop predictive capacity and have revealed little about the ecology of the native fish. Until a more focussed, detailed, and statistically rigorous sampling regime, combined with experimentation and hypothesis testing, is undertaken, recommendations made by ecologists to managers will remain difficult to defend.

These surveys suggest that the impact of invasive fish species on the native fish species surpasses any other. The combined impact of invasion and water abstraction, however, has clearly been devastatingly detrimental. A priority for research in this catchment, as much as elsewhere in South Africa, is to understand how further modifications to freshwater ecosystems could facilitate the spread of invasive species and further enhance their negative impact. In addition, life history and ecological information on the indigenous species is urgently required in order to test existing paradigms regarding the ecological requirements of freshwater fish living in these rivers, and possibly mitigate some of the negative impacts which may accrue as a consequence of human activities. In the Olifants and Doring Rivers, however, recruitment is either non-existent or extremely episodic in all but the uppermost reaches of tributaries. Future studies therefore need to target these reaches.

7.1 RESEARCH PRIORITIES

Specific information needs on the endemic fish of the Olifants and Doring Rivers are listed and described in the following section.

7.1.1. Habitat/flow-mediated invasion

Non-native species appear to be the strongest predictor of the presence/absence of indigenous juvenile fish in the Olifants and Doring Rivers. The continued decline of native species is likely to be a consequence of the spread of invasives through the system, which may be accelerated by altered habitat or flow conditions. Identifying which environmental variables (e.g. flow, cover, temperature) may be responsible for regulating invasive populations would highlight which human activities are likely to be most responsible for range extensions. This information may provide a means of limiting, halting, or even reversing invasion. Because of their overwhelming predominance throughout the catchment, bass *Micropterus* spp. and bluegill sunfish *L. macrochirus* should be prioritised for research.

7.1.2. Migration – effects of river fragmentation

It is believed that fragmentation of migration corridors due to artificial barriers and flow regulation has affected the dispersal, colonisation, migration – and ultimately recruitment – of the large cyprinids in the mainstem of the Olifants River. Despite circumstantial evidence that this has occurred, no studies have examined the role that migration plays in the life history of these indigenous freshwater fish species. This project has developed the methods and cultivated links with international research institutions that will make a study of the effects of river fragmentation possible as soon as funding becomes available.

7.1.3. Flows necessary to maintain spawning habitat

The degradation of spawning habitat (sedimentation, riffle encroachment by marginal vegetation) due to flow regulation (e.g. downstream of Bulshoek Barrage) is believed to have reduced recruitment. Spawning by Clanwilliam yellowfish has been recorded in fast flowing riffles downstream of the Clanwilliam Dam. A comprehensive understanding of spawning habitat is not possible until more spawning sites for the Clanwilliam yellowfish can be located and quantitatively described. Spawning habitat requirements for the Clanwilliam sandfish and sawfin are unknown.

7.1.4. Spawning cues

Modification of temperature and flow in the rivers due to flow regulation is believed to reduce frequency of spawning. An investigation of the temperature and flow conditions required to trigger spawning by Clanwilliam yellowfish downstream of the Clanwilliam Dam was undertaken by Cambray *et al*. (1997) and King (*et al*. 1998). There is a need to corroborate this information with studies from elsewhere in the catchment and extend the study to spawning by sawfin and sandfish.

7.1.5. Larval/post-larval habitat

Fish larvae are known to use shallow littoral areas for predator avoidance and as flow refugia. These areas are sensitive to changes in flow and the survival of larvae is dependent on their hydraulic stability. A clearer understanding of the importance of these areas for local fish species is required.

7.1.6. Flows to maintain river connectivity or exclude invasive species

- *Aerobic/anaerobic swimming capacity*

  Information on the ability of juveniles, sub-adults and adults to overcome hydraulic barriers by swimming is important for exclusion of invasive fish, for design of fishways, and for defining minimum/maximum discharge for migration.

- *Jumping height*

  Information is needed on the ability of juveniles, sub-adults and adults to overcome hydraulic barriers by jumping. This information is important for exclusion of invasive fish and for design of fishways for local species.

7.1.7. Population size estimates

Scientific methods for determining the size of populations need to be advanced in freshwater ecosystems in South Africa. This information is necessary for assessing the conservation status of the species, for monitoring changes in population abundance, and for determining the vulnerability of populations to research interventions such as telemetry.

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